Health cost of land transport noise exposure in New Zealand

October 2024

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NZ Transport Agency Waka Kotahi research report 715
Contracted research organisation – AECOM

NZ TRANSPORT AGENCY WAKA KOTAHI

Te Kāwanatanga o Aotearoa
New Zealand Government

ISBN 978-1-99-106843-9 (electronic)
ISSN 1173-3764 (electronic)

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Evans, L., Wright, J., Rowcroft, P., Jakhu, S., Wagner, L., Hong, T., & Klap, D. (2022). *Health cost of land transport noise exposure in New Zealand* (NZ Transport Agency Waka Kotahi research report 715).

AECOM New Zealand Ltd was contracted by NZ Transport Agency Waka Kotahi in 2020 to carry out this research.

Keywords: cost, GIS, health, modelling, noise.

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Acknowledgements

The authors of this report wish to thank NZ Transport Agency Waka Kotahi (NZTA) for funding this research project into the social health costs of land transport noise exposure throughout New Zealand.

Throughout the development of this report, contributions of the esteemed steering committee chair and members have provided immeasurable value with constructive challenge and guidance. The steering committee comprised the following members:

- Christine Moore, Steering Committee Chair (NZTA)
- Stephen Chiles (on behalf of the Ministry of Health)
- Janet Petersen (NZTA)
- Greg Haldane (NZTA)
- Drew Bingham (Ministry for the Environment)
- Varsha Mala (KiwiRail)
- Sonja Miller (Stats NZ)
- Richard Jackett (NZTA)
- Iain McGlinchy (NZTA)
- Jamie Rodriguez (KiwiRail)

The authors of this report also offer special recognition to peer reviewers who have participated in this research, offering invaluable thought leadership and technical challenge to the research, and reporting prepared.

- Bernard Berry (Independent Consultant, UK)
- Kelvin Norgrove (Strategease)

Contents

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Executive summary

Transport noise has been linked to adverse health outcomes for individuals, including causing annoyance, sleep disturbance, and cardiovascular disease. At an individual level, this leads to costs associated with the impact of noise, including direct costs of medical care and loss of wellbeing. When transport noise from the road and rail network is considered at a national scale, the costs associated with health impacts grow to a significant amount.

The aim of the social cost (health) of land transport noise exposure in New Zealand (SCON) research project was to define the health costs of exposure to transport noise across New Zealand, specifically from New Zealand's road and rail networks.

The key objective of this research project was to establish a methodology for assessing health costs of land transport noise exposure. This included:

- carrying out a literature review to identify existing research in each key field, applicability to the New Zealand context, and gaps in current understanding
- developing an appropriate methodology for predicting and assessing noise from New Zealand's land transport network
- identifying appropriate health indicators and their associated dose-response relationships for land transport noise
- developing and applying a model for quantifying costs associated with noise dose related to each health indicator
- developing the SCON dashboard, an online tool where the results of the research could be viewed, along with a separate web map application to view the results of the noise modelling.

This research project was carried out from October 2020 to August 2022, with input from authors based in New Zealand, Australia, and the United Kingdom.

Health effects considered

The health effects that were investigated for this research project were annoyance, sleep disturbance, and ischaemic heart disease (IHD).

- **Annoyance** – annoyance is one of the most prevalent responses to noise, and it is described as a stress reaction that encompasses a wide range of negative feelings, including disturbance, dissatisfaction, distress, displeasure, irritation, and nuisance.
- **Sleep disturbance** – noise can cause difficulty in falling asleep, awakening and alterations to the depth of sleep, especially a reduction in the proportion of healthy rapid eye movement sleep. Other primary physiological effects induced by noise during sleep can include changes in glucose metabolism and appetite regulation, impaired memory consolidation, and a dysfunction in blood vessels. Long-term sleep disturbance can also lead to cardiovascular health issues. Exposure to night-time noise can also induce aftereffects that can be measured the day following exposure, while the individual is awake, and include increased fatigue, depression, and reduced performance.
- **IHD** – this is caused by the narrowing of coronary arteries, restricting blood flow to the heart. There is evidence that night-time noise is associated with more adverse cardiovascular effects compared to daytime noise. Night-time noise also leads to a stronger stress reaction as indicated by higher neurohormone levels, higher increases in oxidative stress, more pronounced vascular stiffness, and arterial hypertension, as well as perhaps a higher incidence of cardiovascular and metabolic diseases.

Relevant exposure-response functions were adopted for each of these health effects to estimate costs associated with predicted noise exposure levels.

Only these health effects were investigated as there was insufficient evidence of an acceptable quality to establish exposure-response functions for other health endpoints.

Identification of dwellings and population counts per dwelling

Population data used for counts per dwelling were based on 2018 Census data. Results were aggregated by 16 regional council areas, 20 health districts, and 66 territorial authorities.

There are no publicly available datasets identifying whether building outlines correspond to dwellings, or the number of people in each dwelling. Therefore, dwellings and the corresponding number of people in them were estimated using an automated process based on the number of building outlines and address points per land parcel, along with the land use for the parcel.

The output of this processing was a dataset of dwellings, along with the number of people in each dwelling.

Noise modelling prediction method and inputs

Noise modelling was carried out using the Calculation of Road Traffic Noise modelling algorithm implemented in SoundPLAN 8.2 noise prediction software.

Inputs to the noise model were:

- topography as 1 m Light Detection Airborne Radar (LIDAR) data where available (on which approximately 75% of relevant noise receivers across the country were located), integrated with a national Digital Elevation Model from the New Zealand School of Surveying at 15 m resolution
- building outlines as extracted from the Land Information New Zealand data service (last updated August 2020)
- road alignments using traffic data provided by Abley (in turn sourced from a combination of Corelogic and RAMM datasets) and the NZTA. The year of road data varies based on location in the source dataset
- rail alignments using train volumes provided by KiwiRail based on data from 2019.

Descriptors used for evaluating noise dose at each dwelling were L_{den} and L_{night}. The noise doses were calculated by modelling road and rail noise across New Zealand. The $L_{Aeq(24h)}$ noise descriptor was used for noise contours produced to visualise noise emissions.

Cost modelling

A cost model was developed to estimate the social cost of transport noise related to each of the health effects identified above. The model used the outcomes of the noise modelling to estimate the social cost of transport noise, based on the information obtained from the literature review.

Key parameters used to inform the cost model include the:

- population exposed – this is the number of people exposed to transport noise, disaggregated by geographical area and distinguished by transport noise type
- proportion of population affected by transportation noise
- disability weighting – this is a measure of the severity of health impact, which in turn is used to derive Quality-Adjusted Life Year (QALY – the value of a year of life in perfect health). Disability weightings consider impacts on both longevity and quality of life
- health value – this refers to the value of one year of life lived in full health (ie, the value of one QALY).

The cost model separates the costs associated with road and rail noise by key geographical statutory boundaries (ie, regional councils, territorial authorities, health districts) to allow Government to easily identify the areas most impacted by road and rail related noise.

Total cost of land transport noise

The results of the cost modelling suggest that exposure to road and rail transport noise costs the New Zealand economy approximately $654 million a year, as a result of its impact on the health outcomes of the community across both the North and South Island. Most of this impact is driven by road-related noise which accounts for approximately $502 million a year, while rail-related noise accounts for a further $152 million a year.

The value of $654 million a year listed above is based on a central scenario where the disability weights, relative risk, and QALY values adopted were central values recommended in the relevant literature. Lower and higher value estimates were also provided in the literature and were used in sensitivity testing to account for the inherent uncertainty in the outputs of the cost modelling. The results of the sensitivity testing suggest a potential range from −48% to +383% about the central scenario value for the cost from road traffic noise, and a potential range from −47% to +292% about the central scenario value for the cost from rail noise.

Abstract

Transport noise has been linked to adverse health outcomes for individuals, including causing annoyance, sleep disturbance, and cardiovascular disease. In this research project, the social cost of health effects from transport noise caused by New Zealand's road network (state highways and arterial roads) and rail network has been assessed.

A literature review of existing studies about noise modelling, assessment of health impacts from noise, and modelling of costs was carried out to inform the methodology for this study.

The study was carried out by:

- identifying suitable health impacts to consider in the study, based on the reliability and quality of existing research for establishing dose-response relationships. The health impacts chosen were annoyance, sleep disturbance and ischaemic heart disease
- identifying dwellings across the country based on publicly available data, along with estimated population counts in each dwelling
- modelling transport noise from New Zealand's road and rail networks to produce counts of affected population per 1 decibel (dB) noise band in terms of the L_{den} and L_{night} noise descriptors (these are A-weighted noise levels over a time period of 24 hours for L_{den} and from 23:00–07:00 for L_{night}, with penalties applied to account for sensitive times of day), with results aggregated by region, health district, and territorial authority
- determining final costs by applying the dose-response relationships on the proportion of the population affected and multiplying this by monetised QALY (Quality-Adjusted Life Year – value of a full year lived in perfect health) values
- producing an online dashboard to visualise the results of the cost modelling. The online tool can display costs per transport mode (road/rail), region, health district, and territorial authority.

From the estimated population counts in dwellings and the noise modelling, estimates of the number of people exposed to each 1 dB band of noise from road traffic and rail noise were prepared. From this data, the cost modelling derived the final estimates of cost to the New Zealand economy from the road and rail transportation network's noise emissions.

It was found that the total social (health) cost of transport noise from New Zealand's road and rail networks is approximately $654 million per year, when using central values recommended in the relevant literature.

The results of sensitivity testing of the cost model suggest a potential range from −48% to +383% about the central scenario value for the cost from road traffic noise, and a potential range from −47% to +292% about the central scenario value for the cost from rail noise. These percentages were obtained by testing 'low' and 'high' scenarios for disability weights, relative risk, and QALY values derived from the relevant literature.

1 Introduction

1.1 Background

Transport noise has been linked to adverse health outcomes for individuals, including causing annoyance, sleep disturbance, and cardiovascular disease. For individual people, this can lead to costs associated with the impact of noise, including direct costs of medical care and loss of wellbeing, among other factors.

Strategic mapping of transport noise across New Zealand has been carried out previously, although this has not been linked to costs associated with health outcomes. Furthermore, no strategic mapping of noise from New Zealand's rail network has previously been undertaken.

The aim of the social cost (health) of land transport noise exposure in New Zealand (SCON) research project was to estimate the health costs of transport noise exposure across New Zealand, specifically from New Zealand's road and rail networks.

This research expands on the national land transport (road traffic) noise mapping project by incorporating other sources of transportation noise like rail.

The calculated location-specific transportation noise levels are the foundation of the study. This information has been used to derive the current population's noise exposure so that the health and cost impacts across New Zealand can be identified and visualised in an interactive tool. Findings from the study also enable stakeholders to understand what health and social impacts could be expected from a projected increase or decrease in transportation noise exposure.

A study of this scale and complexity is unique for New Zealand. Consequently, much of the methodology has been developed using guidance provided by the European Union (EU) Environmental Noise Directive 2002/49/EC (END).

This research project was carried out from October 2020 to August 2022, with input from authors based in New Zealand, Australia, and the United Kingdom.

1.2 Objectives

The key objective of this research project was to establish a methodology for assessing health costs of transport noise exposure. This included:

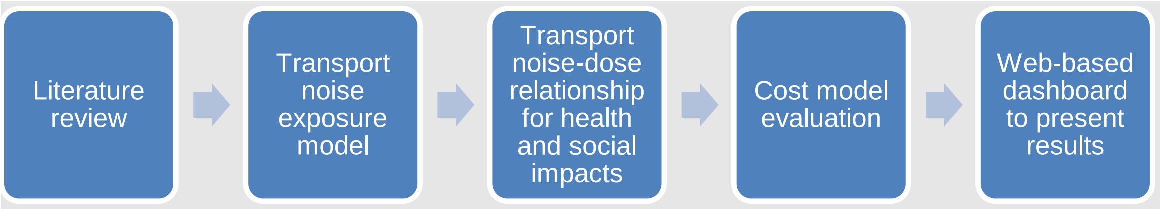
- carrying out a literature review to identify existing research in each key field, applicability to the New Zealand context, and gaps in current understanding
- developing an appropriate methodology for predicting noise from New Zealand's transport network
- identifying appropriate health indicators and their associated dose-response relationships for transport noise
- developing and applying a model for quantifying costs associated with noise dose related to each health indicator
- developing the SCON dashboard, an online tool where the results of the research could be viewed, along with a separate web map application to view the results of the noise modelling.

2 Overview of methodology

2.1 Overall methodology

Figure 2.1 shows the key steps taken in the research project.

Figure 2.1 Key steps of research project



A literature review for the project was undertaken in three parts to focus on the noise modelling, health implications, and social costs of land transport noise exposure in New Zealand. The findings are presented in Appendix A.

2.2 Noise model methodology

2.2.1 Objective

The overall objective of the transport noise exposure model is to determine the social cost of health effects caused by land transport noise exposure on New Zealand's population.

2.2.2 Approach

A three-part literature review was first undertaken to:

- identity existing scientific research and regulatory approaches to transportation noise modelling
- determine the applicability of the findings in a New Zealand context
- confirm that the modelling methods proposed within the SCON project scope and methodology were appropriate for the project
- highlight gaps in understanding and where further work could be undertaken.

The information considered includes:

- the different noise descriptors used to describe transportation noise impacts
- calculation methodologies used locally and abroad for calculating road and rail noise levels
- indicators used to support the quantification of human health and economic impacts from transportation noise in New Zealand
- methods typically used to convert the noise modelling indicators to ensure consistency across different transportation types
- the status of strategic noise mapping and good practice guidelines from the EU.

2.2.3 Noise model development

2.2.3.1 Transportation noise sources

Initially, the transportation noise sources that were agreed to be considered (and their data sources) were the:

- road network (state highways and arterial roads) – Abley
- rail network – KiwiRail
- airports – local aviation authority and regions
- seaports – local port authority and regions.

The project included scope to calculate transport noise from road and rail noise sources based on the available road geometry, transit volumes, speeds, and well-established modelling algorithms.

It was anticipated that transportation noise from airports and seaports would be based on historical noise modelling outputs made available by local authorities. However, the spatial datasets used to create these outputs were not widely available.

One of the reasons for this is the lack of a standardised nationwide approach to predicting and displaying noise from air and seaports. Two New Zealand Standards[1,2] describe the assessment methodology to predict sea and airport noise, however, the regulatory requirement for predictions falls under the framework of the Resource Management Act 1991 (RMA). This can be implemented at a regional or district level through the regional plan, district plan, resource consent, or designation conditions depending on where the port is located. Because of this, there is no centralised source of noise maps for airports and seaports in New Zealand.

Where digital or Portable Document Format (PDF) noise maps of airports and seaports were available, the outputs primarily displayed industry-specific noise contours that are relevant to a specific planning threshold.

For example, seaport and airport noise contour maps would typically show outputs relevant to the following values:

- Seaport: 65 dB L_{dn} 'Inner Control Boundary' (ICB) and a 55 dB L_{dn} 'Outer Control Boundary' (OCB).
- Airport: 65 dB L_{dn} and 55 dB L_{dn} thresholds.

The lack of standardisation also meant that the presentation of this information could vary depending on the local authority, with some maps showing intermediate contours in 1 dB increments, others in 5 dB increments, and others showing the 55 and 65 dB L_{dn} contours only. In any case, the historical seaport and airport noise levels were not available in a format that would allow for values at individual noise-sensitive buildings or property boundaries to be quantified without significant data manipulation.

Without the time and budget required to extract noise levels from the available seaport and airport datasets, the assessment of health effects based on the project methodology needed to be limited to road and rail noise sources.

[1] NZS 6805 Airport noise management and land use planning

[2] NZS 6809 Port noise management and land use planning

2.2.3.2 Project study area

An assessment distance of 600 m either side from road and rail infrastructure boundaries was adopted. This distance was chosen to allow noise levels to be captured in the model at which onset of health effects may occur.

2.2.3.3 Sensitive receptors (Protected Premises and Facilities)

For the purpose of this study, sensitive receptors are defined as per New Zealand Standard 6806:2010 – Acoustics – Road-traffic noise – New and altered roads (ie, Protected Premises and Facilities (PPFs)). These are categorised as:

- buildings used for residential activities including boarding establishments, retirement villages, and temporary accommodation such as hotels and motels
- marae (Māori cultural meeting place)
- spaces within buildings used for overnight patient medical care
- teaching areas and sleeping rooms in buildings used as educational facilities.

This description is consistent with the NZTA mapping guidance. All PPFs within the project study area were identified through an automated process and included with all other buildings in the noise models.

Various methods were tested for the identification of dwellings within a given land parcel. The chosen methodology identified dwellings based on building outlines, parcels, and address points in ArcGIS Pro, through the use of rules or attribute queries. These were also used to estimate the number of people within each dwelling. Further details are provided in section 4.2.2.1.

Building heights were based on Digital Surface Model (DSM) where it was available for New Zealand. Where DSM data was not available, building heights of 6 m for residential buildings (including sheds and garages) and 8 m for commercial and industrial buildings was assumed.

2.2.3.4 Noise source modelling

The most recently available LIDAR data were used to model terrain across all regions of New Zealand. This was supplemented with a national Digital Elevation Model (DEM).

Roads within the transport noise exposure model included state highways, regional and arterial roads. The heights of modelled road centrelines were based on either surveyed information or the processed ground topography.

Rail alignments for existing operational passenger and freight services were provided by KiwiRail. This included main trunk lines, secondary mainlines, and branch lines.

A 3D representation of the existing environment surrounding each of the transport alignments was implemented in SoundPLAN v 8.2. Further details are provided in sections 4.2.5 and 4.2.6.

2.2.3.5 Noise exposure levels

Different indicators are often used to interpret noise depending on the type, duration, and the receptor. Previous studies of human response to noise have resulted in the development of noise indicators in terms of frequency and time weightings. These indicators may be used to approximate human response to noise. This is described further in section 3.

The assessment of population noise exposure was undertaken in consideration of the following noise level indicators, in line with the human health and social cost requirements and New Zealand standards:

- $L_{Aeq(24h)}$
- L_{den} and L_{night}

Transport noise levels were predicted at each dwelling. Façade noise maps and grid noise maps were calculated using the model. The façade noise map results were used to determine the counts of people and dwellings within each 1 dB noise band, which was then entered into the cost model. The grid noise maps were entered into an online Geographic Information System (GIS) map platform for visualisation.

2.2.3.6 Noise exposure tool (SCON dashboard)

The SCON dashboard was configured to allow the user to produce PDF maps based on the view extent filtered by transport mode, region, district or urban areas, or a combination of those. The deliverables for this tool include:

- a basic user guide, explaining the functionality of the tool to the end user, refer Appendix B
- the ArcGIS database and all associated data and tools.

The SCON dashboard includes the overlays (in terms of acoustics) for:

- buildings with free-field noise levels (determined from noise levels incident on the façades of buildings)
- road transport network
- rail transport network
- road noise contours
- rail noise contours
- census data per region.

2.3 Health review methodology

2.3.1 Review objectives

The overall objective of the health review is to determine from available evidence and studies appropriate and robust exposure-response relationships for health, productivity, and cognitive impacts of transport noise.

To achieve this objective the health review follows on from the literature review (presented in Appendix A2), where relevant papers and studies have been identified to further inform the existing understanding on the health effects of transport noise.

More specifically the health review has involved a critical review and evaluation of the literature to identify appropriate and robust health effects (or endpoints) and exposure-response relationships to be used in the quantification of the impacts and costs of transport noise on health.

The approach, described below, provides further detail on how the evidence and studies will be reviewed to determine what is ‘appropriate and robust’.

2.3.2 Approach

A literature review has been undertaken to identify studies relevant to the assessment of health effects from transport noise. The literature review is presented in Appendix A.2.

For all studies identified in the literature review, these have been grouped as follows:

1. Regulatory reviews where health endpoints and exposure-response relationships have been adopted, with the basis for selecting these relationships further discussed.
2. Reviews of published studies, particularly the World Health Organization (WHO) (2018) evaluations, that provide commentary on the limitations or deficiencies of the approaches adopted.
3. Studies that provide supporting information on the biological plausibility of various health effects of transport noise.
4. Epidemiological studies that have not been considered in the WHO (2018) or enHealth (2018) critical reviews. These include cohort studies, case-control studies, cross-sectional studies, and meta-analysis. In relation to the study type it is noted that meta-analysis studies are preferred, with cohort and case-control also preferred particularly where these are longitudinal. Cross-sectional studies are the least preferred, however these are the most common studies used in the assessment of noise annoyance and hence for the assessment of noise annoyance, these studies have a higher initial ranking.
5. Studies specific to New Zealand.

Studies and information identified in points 1 to 3 and point 5 above have been used as supporting information for the identification of health effects where robust information supports a causal or strong association in relation to exposure-response.

Studies identified in point 4 have been critically reviewed to rate the studies in terms of quality (and robustness). Firstly, the studies have been reviewed to identify whether:

- exposure has been defined – this is a noise exposure either modelled or measured in dB
- the source of the noise is defined, and other sources have been excluded
- the study provides sufficient detail on how the outcomes – health measures – have been defined and quantified
- the study defines an exposure-response relationship.

For studies that are not excluded based on the above, the Grading of Recommendations, Assessment, Development and Evaluation (GRADE) system has been used as a tool for the interpretation of the quality of evidence, particularly in relation to study/review outcomes for causality and associations between noise sources and health effects (refer to Appendix C for further detail).

Application of the GRADE system has considered the study limitations, inconsistencies in the study, directness, precision, and publication bias. In addition to these factors, confounders (and how these have been addressed) and the magnitude of the effect identified have also been considered. This system provides certainty ratings ranging from very low to high.

The studies evaluated as detailed above have been used to determine:

- if the health effects identified and evaluated by the WHO (2018) remain relevant, or if sufficient new information supports inclusion of additional health effects in the quantification of health impacts of transport noise
- whether the exposure-response relationships defined by the WHO (2018) remain relevant and robust or appropriate for use, based on outcomes of the more recent reviews and studies
- whether the exposure-response relationships can be refined further in relation to the assessment of different modes of transport, urban versus rural impacts, and if additional considerations should be considered to address the New Zealand population and/or the sensitivities of specific populations.

The above has been used to provide recommendations on the selection of robust exposure-response relationships for the New Zealand context, and how these should be integrated into the noise and cost modelling.

2.3.3 Application of GRADE system

The GRADE system has been adopted as the systematic review method to assess the quality of a body of evidence. The system rates the overall quality of evidence available (ie, all the available studies combined). This is the method that has been adopted by the WHO (Guski et al., 2017), enHealth (2018) and the Department for Environmental, Food and Rural Affairs (UK) (DEFRA) (ARUP, 2020; van Kamp et al., 2020) in reviews of studies relating to the health effects of noise. GRADE has four levels for the quality of evidence ranging from very low to high.

Table 2.1 The levels of quality of evidence of the GRADE system (van Kempen et al., 2018)

Quality of evidence	Definition	Examples
High	Further research is very unlikely to change our confidence in the estimate of effect	Several high-quality studies with consistent results
Moderate	Further research is likely to have an important impact on our confidence in the estimate of effect and may change the estimate	One high-quality study or several studies with some limitations
Low	Further research is very likely to have an important impact on our confidence in the estimate of effect and is likely to change the estimate	One or more studies with severe limitations
Very Low	Any estimate of effect is very uncertain	No direct research evidence One or more studies with very severe limitations

A range of domains were used to appraise the quality of the evidence. Risk of bias is first assessed at the individual study level. The rest are assessed by looking at the entire body of evidence for that outcome. These domains are as follows:

1. **Risk of bias** – assessed at individual study level. It is used to assess limitations with the study and degree of confidence in the findings.
2. **Inconsistency of results** – inconsistency in participants, methodology, and outcomes across the body of evidence. An evaluation of the similarity of point estimates and/or extent of overlap of confidence intervals (CI) may be used.
3. **Indirectness of evidence** – the differences between study characteristics (such as participants, exposures, and outcomes) and those of interest (such as populations of interest) within the body of evidence. The greater the difference, the more indirect the evidence. It may be appropriate to use interchangeably with the terms ‘applicability’ and ‘generalisability’.
4. **Imprecision** – an assessment of 95 percent CIs to ascertain whether the estimate of effect for the body of evidence is sufficiently precise. This is more difficult if CIs are not reported and is generally only used in meta-analysis.
5. **Publication bias** – suspected when evidence comes from a number of small studies, most of which have been commercially funded.
6. **Large magnitude of effect** – presence may justify increasing the rating for the quality of the body of evidence.
7. **Plausible confounding** – which would reduce a demonstrated effect.
8. **Dose-response gradient** – presence may justify rating up the quality of the body of evidence.

The overall quality assessment then involved two main stages.

- First, the risk of bias within each individual study and each individual outcome within the study was assessed, with the approach adopted consistent with that detailed by enHealth (2018).
- Second, the overall quality of the body of evidence for each individual outcome was assessed, as detailed below.

An initial quality level is set by the study design, with randomised control trials considered high quality. However, for the assessment of environmental noise, the exposure is never randomised, hence the following is also applied (ARUP, 2020):

- If any of the evidence is from a longitudinal or intervention study the initial assessment is considered to be high quality.
- If the evidence is only available from cross-sectional studies then the initial assessment is given as low quality.

The GRADE system allows for these initial ratings to be further upgraded or downgraded according to specific criteria. The factors that are used to determine an upgrade or downgrading of the quality of evidence are detailed in Table 2.2.

Table 2.2 Downgrading and upgrading considerations in the GRADE system (ARUP, 2020; WHO, 2018)

Downgrade	Further comments
Study limitations or risk of bias in all studies that make up the body of evidence	If many of the studies have methodological limitations or flaws, or have been rated as having uncertain bias or high bias then the quality of evidence is downgraded
Inconsistency of results between studies	If the findings of the studies are mixed in terms of whether there is an effect, the quality of evidence is downgraded. If there is only one study then consistency cannot be evaluated and the quality of evidence is downgraded
Indirectness of evidence in the studies	If some of the studies are not comparable in terms of population, exposure, comparator, and outcome then the quality of evidence is downgraded
Imprecision of the pooled effect estimate	If the sample size is not large enough to calculate a precise effect estimate, or has a large confidence interval then the quality of evidence is downgraded
Publication bias detected in a body of evidence	This relates to the publication of all findings relevant to the effects, even if those are null findings

Upgrade	Further comments
High magnitude of the pooled effect	If the studies provide large estimates of the magnitude of effects then the quality of evidence is upgraded
Direction of residual confounding and biases opposes an effect (ie, when all plausible confounders are anticipated to reduce the estimated effect and there is still a significant effect)	If the evidence across the studies suggests that all plausible confounders have been accounted for then the evidence is upgraded
Exposure–response gradient	If all the studies provide significant exposure-response relationships (ie, effect increases as dose increases) then the quality of evidence is upgraded

2.4 Cost model methodology

2.4.1 Introduction

Exposure to transport noise is associated with a wide range of adverse impacts on human health, quality of life, wellbeing, public amenity, productivity, and ecosystems. There is evidence linking exposure to persistent or high levels of transport noise with annoyance, sleep disturbance, cardiovascular disease, and impaired cognitive functioning in children (WHO, 2018). There are also increasingly sophisticated ways of quantifying and valuing these impacts in terms of their effects on morbidity and mortality (WHO, 2011; Brown & van Kamp, 2017; Mueller et al., 2017; Van Kempen et al., 2018).

The aim of the cost model is to provide a means for estimating (quantifying and valuing) the social costs (including premature mortality and morbidity) associated with each of a number of health endpoints.

This section outlines the scope of the cost model, and an overview of the general approach methodology used to develop the cost model. A summary of the key modelling assumptions is provided in section 5.2.

2.4.2 Scope

The scope of the cost model in terms of coverage of health and wider social impacts was guided by the findings of the cost model and health impact literature reviews. More specifically, it was driven by the strength of the evidence base with respect to:

- the degree to which it was possible to establish robust and reliable relationships (**exposure-response relationships**) between noise levels for each of road, rail, aviation, and ports and health or other outcomes
- the extent to which reliable and accepted **approaches to quantifying (in physical terms) and valuing (in monetary terms)** the health and wider social costs exist the availability of **datasets** appropriate to the scale and context for which noise estimates were sought.

The **applications or types of decisions** that the cost model and resulting social cost of noise estimates are intended to inform was an important consideration for this study. Examples of possible applications include:

- appraisal of the effects of transport projects
- supporting the case for policy and other measures for the designation or protection of quiet or relatively quiet areas
- gaining a baseline understanding of the burden of disease (at national, regional, or local level) of transport noise exposure (which is the focus of this study) and how this may change over time as a result of noise policy and other measures.

Based on the expected use of the cost model, the following factors were considered:

- The resolution at which the model was needed to operate and therefore at which level data needed to be obtained, recognising that the chosen spatial unit of analysis is also limited by the availability of data. A finer spatial resolution would help decision makers to gain a better insight into health equity issues within cities and identify high risk spots more precisely. However, downscaling data inappropriately would also result in increased uncertainty and/or error in the burden of disease estimations. The SCON model utilised spatial units aligned with the 2018 Census population data to generate results at territorial authority, regional council, and health district level.
- The time horizon over which costs were estimated. For the purposes of this study, the costs were estimated at a single point in time (2022), with the cost representing an annual cost to society. It should be noted that for use in policy appraisals, the estimates would need to be adjusted to take account of inflation and changes in income.

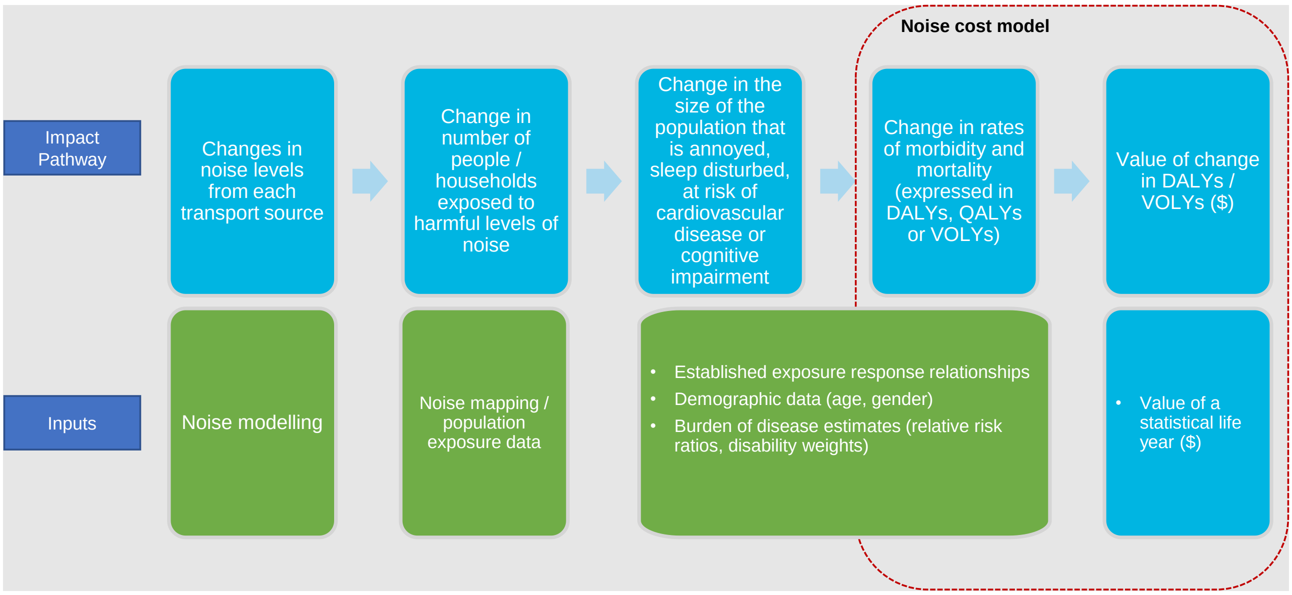
- The extent to which it is necessary to understand, model, and value the cumulative or combined effects arising from exposure to multiple sources of transport noise. Research by Miedema et al. (2003), indicated that the percentage of people who are sleep disturbed as a result of transport noise depends on both the level and source of noise. Thus, even if noise levels measured at a building façade and generated by two or more sources are similar, it should not necessarily be assumed that the level of sleep disturbance caused by the different sources were identical. For this reason, the effects of different noise sources on particular health outcomes were modelled and valued separately. This reflects the fact that the main body of evidence on environmental noise focuses on source-specific impacts of noise on health outcomes and does not incorporate combined exposure effects of multiple noise sources. This also aligns with the approach adopted by the WHO in their latest guidelines (WHO, 2018).

2.4.3 Approach and methodology

The approach to quantifying and valuing the health effects from transportation noise typically follows an impact pathway (see Figure 2.2 for a simplified representation). The impact pathway provides a structured and transparent way of linking the sequence of events between changes in noise levels (eg, as a result of a transport-related intervention) and the outcomes or impacts that can be valued in monetary terms.

The cost model combines information on noise exposure, exposure-response relationships (based on the comprehensive review of epidemiological studies and other relevant evidence), demographic and health data, burden of disease estimates, and relevant cost data. As shown in Figure 2.2, the cost model is reliant on key outputs from the acoustics, health impact, and GIS workstreams.

Figure 2.2 Simplified impact pathway and inputs to cost model



2.4.3.1 Methodology

The development of the cost model involved three broad components:

- The literature review which established the range of approaches that have previously been applied to calculate and value the disease burden from transport noise.
- The assembly and analysis of demographic, health, and economic evidence to support the estimation of the social costs of noise.

- Model building – which involved working closely with the acoustics and GIS workstreams to integrate health and economic evidence with noise and other relevant spatial data.

2.4.3.2 Literature review

The cost model has been developed to be consistent with existing approaches to valuing health impacts of environmental externalities in New Zealand and, in particular, the Health and Air Pollution in New Zealand (HAPINZ 2.0) model/tool which includes an integrated spatial tool/model for exposure and costs. Note that the cost model was developed prior to the updated HAPINZ 3.0 model/tool being released and therefore may not reflect methodological updates made in the latest guidance.

In addition to the HAPINZ study, a range of other tools, models, and guidance were reviewed. A research protocol (see Appendix A3.1) was developed and used to help guide the review and to ensure that as much of the available relevant research from New Zealand and elsewhere was retrieved and considered. It should be noted that a particular focus was given to approaches and methodologies that have practical application and that have been used to support policy- and decision-making elsewhere. Key sources of information in this regard were considered likely to include the WHO, the Organisation for Economic Co-operation and Development (OECD), the European Commission, the UK's Interdepartmental Groups on Costs and Benefits (Noise Subject Group), Public Health England, the UK Department for Transport's Transport Appraisal Guidance (TAG), the UK Independent Commission on Civil Aviation Noise (ICCAN), the UK Civil Aviation Authority (CAA), Australian Transport Assessment and Planning (ATAP) guidance, and Te Tai Ōhanga – The Treasury.

The purpose of the review was to establish which health endpoints should be included in the cost model, subject to the availability of suitably robust approaches and evidence for quantifying and valuing those impacts.

Details of each of the studies retrieved was recorded in a Microsoft Excel database which was structured to allow storage of the full inventory of health impact and social cost/valuation studies reviewed, including meta-data about the study (ie, authors, date, geographical focus, health and other impacts considered, valuation approach, etc) and other relevant information.

2.4.3.3 Data collation

In order to estimate the social costs associated with transport noise, the following data was required:

- The **distribution of transport noise exposure** within the population for each noise source and for relevant intervals (eg, 5 dB) within the ranges relevant to each health endpoint. This data was provided by the acoustics/noise modelling workstream.
- The **exposure-response relationships, relative or absolute risk, or odds ratios** for each noise source and health endpoint.
- **Demographic data** (population size, household size, age, gender) available from Stats NZ.
- **Health (disease and mortality) data** (eg, population-based estimates of the incidence, prevalence, or relative risk of coronary heart disease and related illnesses from surveys or routinely reported statistics). When estimating the impacts of changes in noise levels on cardiovascular diseases (eg, acute myocardial infarction (AMI), hypertension, dementia and stroke), it is necessary to first understand the prevailing risk of these diseases in the affected area as well as the odds ratios which describe the risk of an event (eg, onset of AMI) relative to the risk inherent to another event (eg, the magnitude of the risk of AMI at noise levels of 70 decibels (A-weighted) (dBA) relative to a baseline of 55 dBA, which may differ by age and gender).
- The value of the **disability weight (DW)** for each health endpoint or indicator to support quantification of Disability-Adjusted Life Years (DALYs) or QALYs. The DW is associated with each health condition and

lies on a scale between zero (indicating the health condition is equivalent to full health) and one (indicating the health condition is equivalent to death). Key sources of evidence on relevant DWs include:

- the 'Environmental Noise Guidelines for the European Region' (WHO, 2018) which are based on a systematic review and critical evaluation of studies relating to the health effects of noise
- the evidence reviews underpinning the WHO (2018) guidelines which are published in the International Journal of Environmental Research and Public Health 2017 special issue on 'WHO Noise and Health Evidence Reviews'
- the health statistics and information systems webpages of the WHO which include DWs for burden of disease estimations on a global scale for various recognised medical conditions
- the European Environment Agency's 'Good Practice Guide on Noise Exposure and Potential Health Effects' (EEA, 2010)
- research conducted on behalf of the UK Intergovernmental Group on Costs and Benefits (Noise) on valuing the human health impacts of environmental noise exposure (Berry & Flindell, 2009)
- research published by DEFRA on valuing impacts on sleep disturbance, annoyance, hypertension, productivity, and quiet (DEFRA, 2014).

- The **monetary value associated with each incremental year of life lost or gained**. This is typically measured in terms of the Value of a Statistical Life (VoSL) which is often used by the transport sector for estimating the costs associated with changes in the prevalence of road traffic accidents. Note that while the value of productivity losses and employment income were examined as part of the literature review, they were not included as part of the scope of the cost modelling due to insufficient evidence as to the monetisation of these impacts.
- **The appropriate discount rate** to reflect the present value of future costs.

Table 2.3 provides a summary of the social costs that were investigated and included in the cost model. The extent to which each of the cost items were considered was dependent on the supporting evidence, the availability of robust exposure-response relationships (based on the findings of the health review where applicable), and the confidence with which reliable value estimates were derived.

Table 2.3 Summary of health and other impacts that were considered in the social cost model[3]

Health outcome	Road	Rail	Health and other impacts that were considered in the social cost model
Annoyance	✓	✓	Morbidity (years of life lived in less than full health); OR Impacts on property values
Sleep disturbance	✓	✓	Morbidity (years of life lived in less than full health) Costs of medication GP visits Lost productive time
Acute myocardial infarction (AMI)	✓	?	Mortality from life years lost or premature death Morbidity (years of life lived in less than full health for survivors) Cardiac hospital admissions Lost productive time Emergency room visits GP visits Costs of medication
Hypertension	?	?	Valued in relation to consequent health outcomes: stroke and dementia
Stroke	?	?	Mortality from life years lost or premature death Morbidity (years of life lived in less than full health for survivors) Cardiac hospital admissions Lost productive time Emergency room visits GP visits Costs of medication
Dementia	?	?	Mortality from life years lost or premature death Morbidity (years of life lived in less than full health for survivors) Cardiac hospital admissions Lost productive time Emergency room visits GP visits Costs of medication
Permanent hearing impairment	?	?	Morbidity (years of life lived in less than full health) Costs of medication GP visits Lost productive time
Cognitive impairment in children	?	?	Lost future earnings

2.4.3.4 Model building

The final step in the process was to combine the relevant outputs for each health endpoint and the relevant datasets into the SCON dashboard. Again, this required close coordination with the acoustics and GIS workstreams.

Where uncertainties in the quantification of health impacts and social cost estimates exist, these were highlighted using confidence ratings. Confidence ratings associated with both exposure-response functions and cost estimates were based on the strength of the underlying evidence, for example:

- Low confidence: evidence was partial and significant assumptions were made, so the data provides only order of magnitude estimates.
- Medium confidence: science-based assumptions and published data were used, but there was some uncertainty in combining them, or methodologies considered were new or experimental and subject to revision, resulting in reasonable confidence in the data.
- High confidence: evidence was peer reviewed, assured, or based on published guidance so there was good confidence in using the data.

The cost model includes unit cost or value ranges (where appropriate) that were used for the purposes of sensitivity analysis. This allows the tool user to investigate how the outcomes may change as a result of selecting alternative values within the specified range of unit cost values.

In addition to the ‘base case’ cost model, provision was made for sensitivity testing so that the model user is able to assess the outcomes (in terms of social cost estimates) arising from changes to any underlying assumptions which, in turn, reflect uncertainties in the evidence base.

A summary of sensitivity tests undertaken in the adopted cost model is described in section 5.4.

3 “?” indicates where evidence reviewed in WHO, 2018 is considered weak or absent.

3 Health review

3.1 Introduction

This section provides further review and evaluation of the papers identified in the literature review to inform the assessment of health impacts from transport noise. The starting points for this review are the more recent detailed evaluations of health effects of noise published by enHealth (2018) and the WHO (2018). Based on these reviews, the key health outcomes for which effects are considered to be critical, and for which impacts can be quantified for transport noise sources, relate to annoyance, cardiovascular effects, cognitive impairment, and sleep disturbance. Other health outcomes that were identified by the WHO (2018) as important include adverse birth outcomes, quality of life, wellbeing and mental health, and metabolic outcomes.

The literature review identified a number of reports that relate to regulatory or agency positions on these issues. These have been used to provide additional context to the health outcomes reviewed.

The focus of the health review relates to studies that have specifically assessed the effect of transport noise exposure on various health outcomes, and detailed reviews that have considered these studies (including current systematic reviews and meta-analysis). The review presented in this section has been undertaken following the methodology outlined in section 2.3.

It is noted that research carried out into health outcomes from aircraft and ports is included in this section. However, these were not considered further in this study due to limitations in available data for noise modelling. This is discussed further in section 2.2. This in turn meant that cognitive impairment was not considered further in this study as there was insufficient evidence to assess cognitive impairment for other noise sources besides aircraft noise.

3.2 Regulatory or agency evaluations

3.2.1 UK

DEFRA (ARUP, 2020; Clark et al., 2020; van Kamp et al., 2020; van Kamp et al., 2019) commissioned two reviews of the published evidence in relation to the health effects of noise, available following publication of the WHO (2018) review.

The first DEFRA review (ARUP, 2020; Clark et al., 2020) was undertaken as a systematic review using the GRADE system to determine if any of the new studies change the WHO (2018) review and outcomes. The DEFRA review considered studies published to March 2019 and addressed the following health outcomes: cognition; dementia and other neurodegenerative diseases, mental health, quality of life and wellbeing, birth and reproductive outcomes, and cancer. Overall, the review determined the following (ARUP, 2020; Clark et al., 2020; UK CAA, 2020):

- Many of the conclusions from WHO (2018) remain unchanged.
- Some of the evidence for road traffic noise and railway noise has increased since 2018, specifically the presence of low-quality evidence for road traffic noise effects on medication use and depression and anxiety as measured by interviews.
- There is low-quality evidence for an effect of road traffic noise, aircraft noise, and railway noise on some cancer (previously no evidence available).

The second DEFRA review was conducted by van Kamp et al. from the National Institute for Public Health and the Environment (RIVM) (van Kamp et al., 2020; van Kamp et al., 2019) and evaluated evidence relating to environmental noise exposure and annoyance, sleep disturbance, cardiovascular, and metabolic health outcomes between 2014 and the end of 2019. While this review was systematic, it did not include use of the GRADE system. This review determined the following:

- Results showed that since 2014, an impressive number of articles have been published addressing the association between transport related noise exposure and the health effects evaluated.
- The number and size of the new studies warrant new meta-analyses, in particular where the cardiovascular effects are concerned, but also for annoyance and sleep disturbance (particularly in relation to aircraft noise).
- For the cardiovascular and metabolic effects, the more recent meta-analysis by Vienneau et al. (2019) should be taken into account.

The Independent Commission on Civil Aviation Noise (ICCAN) commissioned the National Centre for Social Research (NatCen) to conduct a rapid evidence assessment to systematically review existing evidence in relation to aircraft noise and health from the WHO and DEFRA reviews, as well as any other evidence subsequently published. This review considered the GRADE system. The key outcomes from the review are as follows (Grollman et al., 2020; ICCAN, 2020):

- For studies relevant to the assessment of birth and reproductive outcomes, diabetes, hypertension, some aspects of sleep, and wellbeing, the evidence of effects of aviation noise were determined to be of low or very low quality under the GRADE system, indicating the need for further research. There was little to no evidence for some areas including dementia and neurodegenerative outcomes, auto-immune disorders, and other cancers.
- For reading comprehension and stroke incidence there is moderate quality of evidence.
- Selected outcomes should be prioritised for further research in the short to medium term, namely sleep, diabetes, wellbeing, depression, and anxiety. A strategy for research in the short to long term should be developed. A range of study designs are required to build the evidence base.

3.2.2 Europe

The most recent review of the health impacts of environmental noise in Europe (EEA, 2020) adopted the WHO (2018) evaluation and exposure-response relationships recommended by the WHO (2018) to calculate population health impacts and burdens.

3.2.3 Australia

The most current review of the health effects of noise is the review completed by enHealth (2018).

3.2.4 New Zealand

A review of the evidence for health impacts of transport in New Zealand was published in 2002 (Kjellstrom & Hill, 2002). This review provided a short summary of information related to the health effects of transport noise with some reference to New Zealand-specific information. This review notes the limited data available that is specific to New Zealand. In relation to road transport, the review notes the following:

- Transit New Zealand evaluated a residential exposure study in Christchurch during the 1990s, which determined the recorded levels of noise were similar to other urban populations. Transit New Zealand established guidelines for maximum daytime noise levels (55 dBA averaged over 24 hours) in 1994.
- Transfund New Zealand evaluated community perceptions of noise and methods to reduce noise from roads.

- Two other studies were being conducted at the time of the report which included a review of noise levels near different road surfaces and annoyance as well as community impacts of noise, specifically annoyance and sleep disturbance.

The second of the two studies noted above was published by Land Transport New Zealand in 2006 (Dravitzki et al., 2006). This study involved a community survey of 138 people in relation to annoyance from road traffic noise. The survey found moderate to strong correlations between a change in noise and change in annoyance, as well as behavioural change (such as closing windows, raising the voice, or changing schedules to avoid noise).

From 2010 onwards, some additional smaller/limited studies/reviews specific to New Zealand have been published. While these are too small to be able to inform the more detailed review presented in sections 3.3 to 3.8, they provide some context to the discussion. These studies include the following:

- Two studies are available that relate to noise levels and health related quality of life (HRQOL). The first study related to 105 participants residing close to Auckland International Airport and also evaluated noise sensitivity (Shepherd et al., 2010). The study found that noise sensitivity was associated with annoyance and HRQOL. Another study evaluated a range of areas – from noisy city areas to quiet rural areas (Shepherd et al., 2013). The study involved 823 participants and identified that quiet areas afforded greater HRQOL than noisy areas.
- A review of health impacts of road transport on the New Zealand population included consideration of impacts from road traffic noise (Briggs et al., 2015). This review incorporated exposure-response functions for cardiovascular effects (stroke, ischaemic heart disease, and hypertension) from a published study (Vienneau et al., 2015) with no detailed review or information on why these relationships were adopted.
- AECOM conducted a noise annoyance study related to road and rail noise for the NZTA (Humpheson & Wareing, 2019). The study was conducted in Auckland and involved 801 participants. This study determined that road traffic was rated most annoying, with the percentage of highly annoyed (%HA) similar to other studies. The study did identify that the onset of annoyance occurred at a marginally lower noise level than other studies suggesting the potential for some level of increased sensitivity to transport noise in the community.
- A study on noise sensitivity in New Zealand focused on potential predictors of noise sensitivity (social and cultural determinants) (Shepherd et al., 2020). The study involved 746 participants and was not specific to transport noise. The study found approximately 50% and 10% of the participants reported being moderately or very noise sensitive respectively, with the key predictors of sensitivity being age, length of residence, level of social deprivation, and self-reported illness.
- The study by Welch et al. (2018) involved a small number of participants (57 in 2012 and 65 in 2015) living near Wellington airport and a further group of participants living away from the airport (control location). The study, which collected data in 2012 and 2015, evaluated noise sensitivity and self-rated health related quality of life. This found that noise-sensitive individuals had poorer HRQOL when living in noise-exposed areas near the airport, compared with those in other areas.

3.3 Annoyance

3.3.1 Overview of adverse effect

Annoyance is a feeling of displeasure associated with any agent or condition known or believed by an individual or group to adversely affect them. It is one of the most prevalent responses to noise, and it is described as a stress reaction that encompasses a wide range of negative feelings, including disturbance, dissatisfaction, distress, displeasure, irritation, and nuisance. The individual response to noise depends not only on exposure levels but also on contextual, situational, and personal factors. It can initiate physiological stress reactions that, if long-term, could trigger the development of cardiovascular disease.

Annoyance levels can be reliably measured by means of an ISO 15666 defined questionnaire, which has enabled the identification of relationships between annoyance and noise sources. There is evidence of sufficient strength for environmental noise annoyance (van Kamp et al., 2020). Exposure-response relationships have been established for noise annoyance from transport sources, including aircraft noise, road traffic noise, and rail noise. The measure of the percentage of the population highly annoyed (%HA) to levels of noise reported as L_{den} (ie, average noise level over a 24-hour period) is considered to be the most appropriate health outcome for evaluating and quantifying effects from noise exposure.

3.3.2 WHO evaluation

The WHO evaluation on annoyance from environmental noise (Guski et al., 2017) evaluated studies published between 2000 and 2014. The studies were systematically reviewed using the GRADE system. The review identified 62 studies of which 46 were used in a quantitative meta-analysis. None of the studies were from populations in New Zealand or Australia. In relation to transport noise, the analysis involved the following:

- Aircraft noise: 15 studies were identified, with data from 12 studies pooled in the meta-analysis, aggregating data from 17,094 participants, with an exposure-response function determined in relation to %HA by L_{den} noise. The quality of the evidence for an association between aircraft noise and %HA was judged to be moderate.
- Road traffic noise: 26 studies were identified, with data from 17 studies pooled in the meta-analysis, aggregating data from 34,122 participants, with an exposure-response function determined in relation to %HA by L_{den} noise. The quality of the evidence for an association between road traffic noise and %HA was judged to be moderate.
- Rail traffic noise: 11 studies were identified, with data from 10 studies pooled in the meta-analysis, aggregating data from 10,970 participants, with an exposure-response function determined in relation to %HA by L_{den} noise. The quality of the evidence for an association between rail traffic noise and %HA was judged to be moderate.

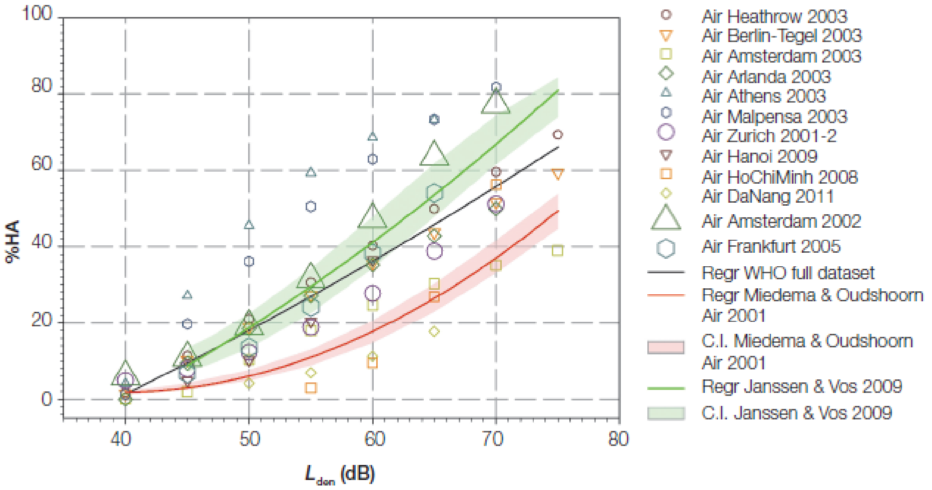
Several research gaps were identified in the review, with the variance in the characterisation of exposure and the measurement and ascertainment of %HA as the main sources of heterogeneity.

3.3.3 Further discussion on WHO evaluation of aircraft noise annoyance

Following publication of the WHO (2018) review and supporting publications, there has been ongoing debate on the approach used for assessing and determining exposure-response functions published by Guski et al. (2017) in relation to annoyance from aircraft noise.

Figure 3.1 presents a summary of the data points from each of the 12 studies evaluated in the meta-analysis by Guski et al. (2017), along with the aggregated exposure-response function from the meta-analysis (labelled 'Regr WHO fill dataset'), with comparison against the exposure-response functions used by regulatory agencies prior to the WHO (2018) evaluation, namely the relationships from Miedema and Oudshoorn (2001) and Janssen and Vos (2009). It is clear from this figure that the exposure-response relationship developed by the WHO (2018) is different to the relationship from Miedema and Oudshoorn (2001), with the WHO suggesting a greater %HA at the same level of noise exposure.

Figure 3.1 Scatterplot of response data from 12 studies included in WHO review and exposure-response function determined from WHO meta-analysis, with comparison against former exposure-response functions (the size of the markers correspond to the number of respondents in each study) (Guski et al., 2017)



In 2018, Gjestland (2018) presented a systematic review of all the evidence relevant to assessing noise annoyance from aircraft noise, not only the studies evaluated by Guski et al. (2017) post-2000. Of key concern is the significant variability on %HA responses in the range L_{den} 50 to 60 dB, which ranged from approximately 5% to 70%, which results in establishing a decreased criteria of L_{den} = 45 dB to prevent adverse health outcomes from noise annoyance from aircraft.

Review of the studies by Gjestland questions the:

- selection of studies
- inclusion of studies from two airports that are not considered representative of airports in general
- use of response weighting in the meta-analysis
- bias towards the inclusion of high-rate change airports
- use of the community tolerance level (CTL) approach
- inclusion of data from surveys not conducted using standardised methods.

Gjestland (2018) presents an evaluation of 18 post-2000 studies that demonstrate that the exposure-response curve is not significantly different to the previous relationships used in the EU (Miedema & Oudshoorn, 2001; Miedema & Vos, 1998) and the lower threshold for effects should be higher at 53.4 dB. The paper concludes that the WHO review is based on questionable evidence.

Guski et al. (2019) published a reply to the Gjestland (2018) critique. This paper addresses each of the issues raised to substantiate the decisions made in the WHO review, refuting the outcomes of the Gjestland (2018) review.

Gjestland (2019b) further replied to Guski et al. (2019) refuting the responses provided and re-affirming the issues raised and conclusions reached in Gjestland (2018).

A more detailed review of exposure-response functions from 65 studies was conducted by Gjestland (2020). This determined that the exposure-response functions pre- and post-2000 are not meaningfully different, particularly in the lower part of the exposure range, and that the previous relationships used in the EU

(Miedema & Oudshoorn, 2001; Miedema & Vos, 1998) remain the best estimate for the prevalence of annoyance from aircraft noise, with a threshold to avoid adverse effects of L_{den} = 54 dB.

Brink (2020) provides a comment on the Gjestland (2020) review, critically reviewing the paper and concluding that the article lacks scientific rigor, in particular the selection criteria for studies, and there is no assessment of study quality. Brink noted that a number of the studies are old and were conducted before well-established questionnaire and scales were used. Gjestland (2021) has further replied to the review by Brink, refuting the comments from Brink. The debate is ongoing and may result in revision of exposure-response relationships for aircraft noise annoyance at some point in the future if, and when agreement between experts is achieved.

3.3.4 DEFRA evaluation

The evaluation commissioned by DEFRA (van Kamp et al., 2020) identified an additional 39 studies that addressed noise annoyance in the period 2015 to 2019, of which 12 studies provided a quantitative evaluation and were ranked as moderate to high quality.

The studies were evaluated using the GRADE system and included the following:

- Road traffic noise: four studies with bias risk rankings of low (one study) to medium (three studies) (Banerjee, 2013; Bunnakrid et al., 2017; Camusso & Pronello, 2016; Ragettli et al., 2015).
- Air traffic noise: three studies with bias risk rankings of low (one study) to medium (two studies) (Bartels et al., 2018; Cho et al., 2014; Quehl et al., 2017).
- Rail traffic noise: two studies with bias risk rankings of low (one study) to medium (one study) (Licitra et al., 2016; Pennig & Schady, 2014).
- Combined sources that include transport sources: three studies with bias risk rankings of low (one study) to medium (two studies) (Brink et al., 2019a; Nguyen et al., 2016; Sung et al., 2016).

The review determined that the additional results warrant a new update of the meta-analysis on annoyance for all transport noise sources: aircraft, road, and rail. The most significant differences in the studies relate to the assessment of aircraft noise, where new data from the DEBATS (France) and NORAH (Germany) studies suggest the WHO evaluation should be updated. The review also acknowledges the current debate described in section 4.3.3 about the validity of the evidence (in particular some studies have not used standardised methods) and review presented in the WHO review by Guski et al. (2017). Hence, any update of the review in relation to aircraft noise annoyance would also require close examination of the studies to be included in the meta-analysis.

However, there are no large differences expected in outcomes from road and rail noise annoyance, and there are no issues raised in the literature in relation to the validity of the studies evaluated for these noise sources. The review completed by van Kamp et al. (2020) does not include an updated meta-analysis.

3.3.5 Additional studies

The literature review conducted for this assessment identified 34 studies published in the period 2015 to March 2021, three of which were identified and systematically reviewed as part of the DEFRA evaluation (Bartels et al., 2018; Brink et al., 2019a; Sung et al., 2016).

Of these studies, 22 were further excluded on the basis that they were duplicates and did not provide new study data, did not quantitatively assess annoyance on the basis of standardised measure of %HA (Badihian et al., 2020; Baudin et al., 2020), and there were no exposure-response relationships evaluated. However, a number of these excluded studies provide qualitative evaluations that assist in understanding relationships between noise sources, annoyance, and other factors, as discussed below.

A number of the excluded studies have not specifically characterised noise annoyance, however they have used noise annoyance as a review indicator for health effects. These reviews have identified that noise annoyance may influence the occurrence of respiratory symptoms (Eze et al., 2018), a significant association between annoyance from night-time noise and a fair/poor self-rated health status (Baudin et al., 2021), as well as an increased risk of hypertension (Baudin et al., 2020). In relation to mental health, the studies provide mixed outcomes with some studies showing either only a weak association or a negative association between transport noise annoyance and mental health (Cerletti et al., 2020; Stansfeld et al., 2021). Other studies suggest an association between those with higher levels of noise sensitivity and poor mental health and high perceived stress levels (not observed in those with higher education) (Jensen et al., 2018). The same study showed a clear association between noise annoyance and perceived stress. The New Zealand study (Shepherd et al., 2020) found traffic noise to be inversely associated with psychological wellbeing.

A study by Siebler et al. (2018) showed that contextual factors (psychological, economic, and social factors) are relevant for the assessment of noise sensitivity and noise annoyance. This is particularly relevant to communities that live in different ways (eg, informal settlements in South Africa) than the populations included in most noise studies which are predominantly urban environments in western cities.

Noise annoyance has also been found to have a negative effect on overall residential satisfaction in urban areas for adults and children (Grelat et al., 2016).

Following this review, eight papers were identified that required further review in relation to quality. These relate to aircraft noise (two studies), road traffic noise (two studies), and combined road, rail, and aircraft traffic noise (four studies). These have been evaluated following the GRADE system.

Appendix A presents a summary of the eight studies evaluated, along with the risk of bias ranking. The papers were predominantly considered to be of moderate to high quality (low to moderate risk of bias) with one study of low quality (high risk of bias).

The papers reviewed by DEFRA (van Kamp et al., 2020), as well as the additional eight studies reviewed in this assessment, have been further considered in conjunction with the GRADE reviews completed in the WHO review (Guski et al., 2017). The GRADE system has then been used to review the overall quality of evidence as assessed by Guski et al. (2017). These reviews are included in Appendix C and show that the additional studies do not change the overall quality of evidence in relation to annoyance from that presented in the WHO (2018) review.

3.3.6 Overall evaluation and recommendations

Since publication of the WHO (2018) review, a number of additional papers have been published that are considered to be of moderate to high quality. These papers add to the existing evidence and the quality of evidence for an exposure-response function between L_{den} and %HA for aircraft, road, and rail noise remains unchanged as medium.

The exposure-response function developed in the WHO (2018) (Guski et al., 2017) review for aircraft noise has been the subject of considerable debate in the literature, with critiques suggesting that the former exposure-response function adopted by the EU (Miedema & Oudshoorn, 2001; Miedema & Vos, 1998) should be retained in preference. Given the new studies that are available – many of which indicate exposure-response relationships between these two curves – it is reasonable that an updated meta-analysis be undertaken to incorporate all suitable studies to update the exposure-response function adopted for aircraft noise. This has yet to be undertaken.

In the absence of an updated relationship for aircraft noise, the relationships established by Guski et al. (2017) would be suitable for this assessment. A sensitivity analysis should also consider the former EU relationship (Miedema & Oudshoorn, 2001; Miedema & Vos, 1998).

The assessment of road and rail noise should adopt the relationships established by Guski et al. (2017).

The recommended exposure-response functions are summarised in Table 3.1.

Table 3.1 Recommended exposure-response functions for quantification of annoyance as %HA#

Health outcome	Noise metric	Lowest level of exposure (dB)	Exposure-response relationship per 10 dB increase (RR= relative risk or OR = odds ratio) [95% confidence interval] Regression equation	Quality of evidence (all studies to 2021) and reference
Road traffic noise	L_{den}	40	OR = 3.03 [2.59–3.55] %HA = 78.9270–3.1162 × L_{den} + 0.0342 × L_{den}^2	Moderate (Guski et al., 2017)
Railway noise	L_{den}	34	OR = 3.53 [2.83–4.39] %HA = 38.1596–2.05538 × L_{den} + 0.0285 × L_{den}^2	Moderate (Guski et al., 2017)
Aircraft noise	L_{den}	33	OR = 4.78 [2.28–10.05] %HA = −50.9693 + 1.0168 × L_{den} + 0.0072 × L_{den}^2	Moderate (Guski et al., 2017)
Aircraft noise – sensitivity analysis	L_{den}	33	%HA = −9.199*10^{-5} (L_{den}-42)^3 + 3.932*10^{-2} (L_{den}-42)^2 + 0.2939 (L_{den}-42)	N/A (Miedema & Oudshoorn, 2001; Miedema & Vos, 1998)

Relationships identified relate to exposures by adult populations

For annoyance, which is considered a less serious health effect than self-reported sleep disturbance, the relevant risk has been determined by the WHO (2018) to be 10%HA. This means the absolute risk associated with exposure should be close to, but not above 10%HA, to be health protective.

3.4 Cardiovascular effects

3.4.1 Overview of adverse effect

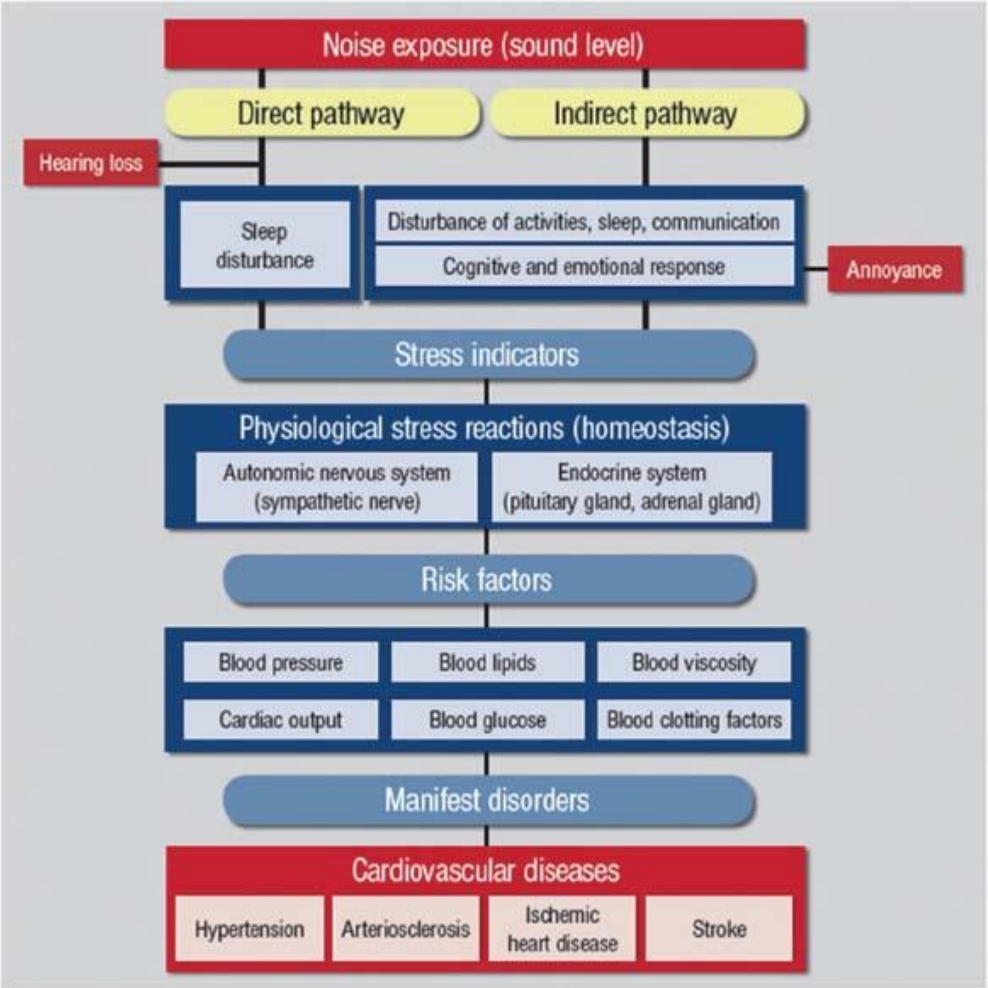
Noise is an important risk factor for chronic diseases. Noise exposure activates stress reactions in the body, leading to increases in blood pressure, a changing heart rate, and a release of stress hormones.

Cardiovascular diseases are the class of diseases that involve the heart or blood vessels, both arteries and veins. These diseases can be separated by end target organ and health outcomes. Strokes reflecting cerebrovascular events and ischaemic heart disease (IHD) or coronary heart disease (CHD) are the most common representation of cardiovascular disease.

High-quality epidemiological evidence on cardiovascular and metabolic effects of environmental noise indicates that exposure to environmental noise, including transport noise increases the risk of IHD.

A link between noise and hypertension is relatively well established in the relevant literature. Whilst there is not a consensus on the precise causal link between the two, there are a number of credible hypotheses. A leading hypothesis is that exposure to noise could lead to triggering of the nervous system (autonomic) and endocrine system which may lead to increases in blood pressure, changes in heart rate, and the release of stress hormones. Depending on the level of exposure to excess noise, the duration of the exposure, and certain attributes of the person exposed, this can cause an imbalance in the person's normal state (including blood pressure and heart rate), which may make a person hypertensive (consistently increased blood pressure) which can then lead to other cardiovascular diseases (DEFRA, 2014). This hypothesis is illustrated in Figure 3.2.

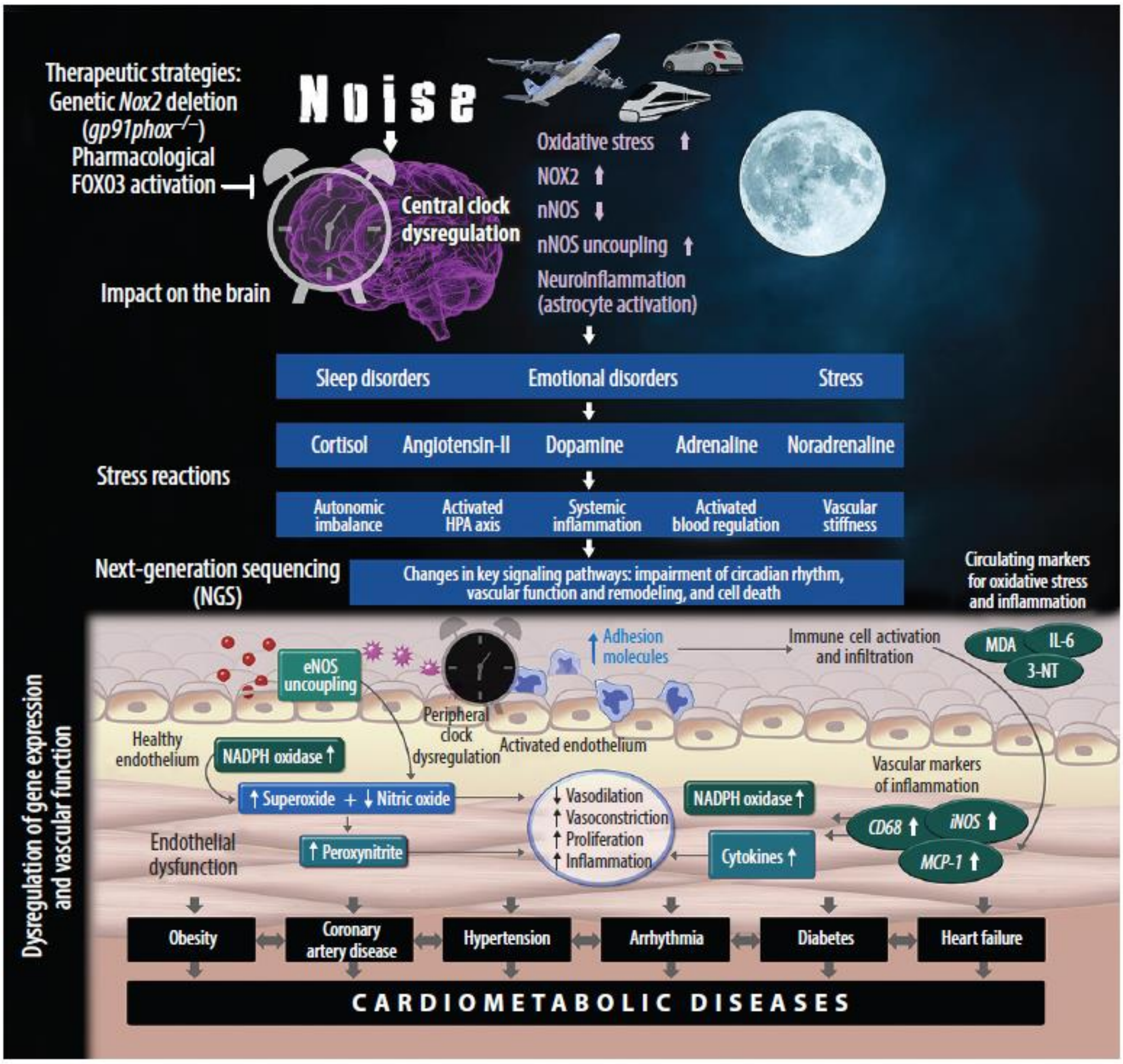
Figure 3.2 Noise reaction model/hypothesis (Babisch, 2014)



A more recent review (Münzel et al., 2020) of evidence provided from epidemiological, translational, and basic science models shows that night-time noise compared with daytime noise is associated with more adverse cardiovascular effects. Compared with daytime noise, night-time noise leads to a stronger stress reaction as indicated by higher neurohormone levels, higher increases in oxidative stress, more pronounced vascular stiffness, and arterial hypertension, as well as perhaps a higher incidence of cardiovascular and metabolic diseases (refer to Figure 3.3). Also, some evidence suggests that intermittent noise with peaks clearly above the background levels (as an intermittency ratio (IR) (Wunderli et al., 2016)) during the night-time may be particularly harmful, with the associations with cardiovascular mortality stronger with moderate IR levels during the night-time (Héritier et al., 2017; Münzel et al., 2020).

Animal models provide some insight on the mechanisms behind the effects of night-time noise on cardiovascular disease (CVD), including a disturbance of the circadian clock (body's internal clock) due to downregulation of genes responsible for regulating the circadian rhythm (24-hour cycles which are part of the body's internal clock). Furthermore, animals exposed to noise revealed significant changes in the expression of genes responsible for the regulation of vascular function, vascular remodelling, and cell death (Münzel et al., 2020).

Figure 3.3 Pathophysiology of night-time noise-induced cardiovascular and brain disease



Genetic *Nox2* deficiency and pharmacological FOXO3 activation by bepridil prevented the adverse noise effects. Abbreviations: 3-NT, 3-nitrotyrosine; CD68, macrosialin; eNOS, endothelial nitric oxide synthase; HPA, hypothalamic-pituitaryadrenal; IL-6, interleukin 6; iNOS, inducible nitric oxide synthase; MCP-1, monocyte chemoattractant protein-1; MDA, malondialdehyde (Münzel et al., 2020).

3.4.2 WHO evaluation

The WHO (2018) review of evidence related to cardiovascular outcomes was completed by van Kempen et al. (2018). This review identified 61 studies covering the period January 2000 to August 2015 of which 53 were used in the quantitative meta-analysis. The review addressed key health outcomes relevant to cardiovascular effects.

3.4.2.1 Hypertension (incidence)

Thirty-seven studies were identified, with the most (27 studies) related to road traffic noise.

The quality of the evidence was evaluated to be very low, with many of the studies showing inconsistent results and should only be considered to be supporting an association between traffic noise and hypertension. This is particularly relevant for the assessment of rail traffic noise where the studies reviewed showed inconsistent results, with either no association or no statistically significant association reported.

Estimates of effects adopted for road and aircraft traffic noise should be considered uncertain.

3.4.2.2 IHD (incidence and mortality)

Twenty-two studies were identified.

For air traffic noise, the only statistically significant association related to the incidence of IHD based on two ecological studies supported by two cross-sectional studies on the prevalence of IHD, with the quality of the evidence evaluated to be very low to low.

The strongest association and most robust exposure-response relationship was identified for road traffic noise and the incidence of IHD. The quality of evidence was evaluated to be high based on three cohort studies, five case-control studies, and one ecological study for the incidence of IHD and seven cross-sectional studies for the prevalence of IHD.

For rail traffic noise there were few studies identified, in which no significant association of effect was identified. The quality of the evidence was evaluated to be very low.

3.4.2.3 Stroke (incidence and mortality)

Fourteen studies were identified, with only one of these relating to rail noise.

For road traffic noise, limited studies were available: two cross-sectional studies relating to the prevalence of stroke, one cohort study on the incidence of stroke, and three cohort studies on mortality due to stroke. The quality of evidence ranged from very low to moderate, however for most effects there was no significant exposure-response relationship identified.

For aircraft noise, limited studies were available: two ecological studies relating to the prevalence of stroke, two cross-sectional studies on the incidence of stroke, and one cohort study and two ecological studies on mortality due to stroke. The quality of all evidence was very low.

For rail traffic noise there was no evidence on the relationship with incidence of or mortality from stroke. One study found an association with prevalence of stroke however this was not statistically significant.

3.4.2.4 Blood pressure in children

The WHO evaluation also considered the impact of noise on children’s blood pressure. In relation to road traffic noise, the studies reported mixed outcomes and there were no statistically significant exposure-response relationships identified. In relation to aircraft noise, two cross-sectional studies were available (including one from Australia), which reported inconsistent results, with the quality of evidence considered to be low quality.

No data was available to assess the impact of noise on children’s blood pressure from rail noise exposures.

3.4.3 DEFRA evaluation

The evaluation commissioned by DEFRA (van Kamp et al., 2020; van Kamp et al., 2019) identified an additional 30 studies on the effects of noise on the cardiovascular system in the period 2015 to March 2019. Some of these studies included data that was already considered in the WHO (2018) review, however not all the data presented in these studies was incorporated. These studies were not reviewed using the GRADE system but were considered in conjunction with the existing information from the WHO.

In the WHO evidence review (van Kempen et al., 2018), despite the quality of evidence being evaluated as very low, hypertension is included as an endpoint, since WHO considered it as one of the critical endpoints for deriving their noise guideline values.

DEFRA had previously reviewed the methodology for conducting health impact assessments (HIA) in a report for the EU commission. This concluded that in the context of undertaking a HIA, hypertension is not necessarily a good endpoint, since it might lead to double-counting of cardiovascular effects in a population (van Kamp et al., 2019). The most recent assessment of the impact of noise in Europe has excluded hypertension as a health endpoint (EEA, 2020).

The DEFRA review of the additional studies/publications suggests that the WHO relationships may require updating for IHD associated with road and air sources (re-run meta-analysis with new studies included). For stroke, a new meta-analysis is suggested for road traffic and aircraft.

For road traffic noise and hypertension, the review suggests a systematic evaluation and possible meta-analysis is undertaken to find out how the conclusions of the WHO evidence review change.

3.4.4 Additional studies

The literature review conducted for this assessment identified 47 studies published in the period 2015 to March 2021. It is noted that an additional two studies on road traffic noise were further identified to October 2021 that have been included in this assessment. These included nine that were identified in the DEFRA review. The DEFRA review also incorporated data from studies previously excluded in this review, due to the studies being confounded with air pollution exposures. For completeness, where these studies were utilised in the DEFRA review, they have been further considered in this review.

Of the additional studies identified in this review, 17 were further excluded on the basis that they were duplicates and did not provide new study data, assessed population health burdens using existing exposure-response relationships, were evaluated on the basis of noise annoyance (or other parameters) and not noise exposure levels, or there were no exposure-response relationships evaluated.

Most of the studies related to the key health outcomes related to cardiovascular effects from environmental noise, namely hypertension, IHD, and stroke. However, some studies evaluated other health outcomes that are summarised below.

Acute myocardial infarction (AMI) and congestive heart failure (CHF) (Bai et al., 2020) where chronic exposure to road noise traffic was found to be associated with elevated risks for AMI and CHF incidence with nearly linear exposure-response relationships. This study involved a large number of participants in Canada (37,441 for AMI cohort and 986,295 for CHF cohort). An additional study from the Danish Nurse Cohort (22,189 participants aged 44 years and older) also identified an association between road traffic noise and the incidence of heart failure (HF) (Lim et al., 2021).

The relationship for AMI is in line with the WHO review of other published meta-analysis evaluations (van Kempen et al., 2018; Vienneau et al., 2015). While these studies support the current outcomes evaluated in the WHO review, the evidence specific to AMI and CHF is limited and insufficient data (from these and two other studies) is available to evaluate AMI and CHF as specific health outcomes for cardiovascular disease. Specifically in relation to CHF and HF, the available studies show mixed results of a statistically significant association. In the study from Lim et al. (2021) the association was attenuated (reduced) following adjustment for co-exposures to air pollution, such as nitrogen dioxide.

Atrial Fibrillation (AF) (Monrad et al., 2016) where exposure to residential road traffic noise may be associated with a higher risk of AF, however associations were difficult to separate from exposure to air pollution. Where corrected for air pollution no statistically significant associations were identified.

Atrial stiffness (Foraster et al., 2017) where an association between long-term noise exposure from transport and a marker for atrial stiffness was identified. Another study looked at carotid intima-media thickness (cIMT) that reflects a change in the vascular walls due to plaque formation (Halonen et al., 2017). A positive but non-significant association was observed between atrial stiffness and road L_{den} levels in urban areas, whereas this association tended to be negative in rural areas. In relation to cIMT, no association was identified after adjustments for confounders (including co-exposure to traffic-related air pollution, education, smoking status, alcohol consumption, body mass index (BMI), physical activity, consumption of fruit and vegetables).

Prevalence of coronary artery disease (CAD) (Gilani & Mir, 2021) was evaluated in 909 adult participants living in areas of India impacted by road transport noise, focusing on exposure to noise levels above 60 dBA as L_{den} comparing outcomes to participants in quiet areas. The study found exposures to traffic noise above 60 dBA was significantly associated with the prevalence of CAD, with the association being strongest in males, particularly older males reporting higher levels of stress and poor sleep quality. This study was small and did not account for air pollution, hence it should only be used as supporting information on the potential impact of traffic noise on susceptible individuals.

Heart rate (Nassur et al., 2019b) was evaluated in a small study (93 participants) in the vicinity of an airport and found that exposure to the maximum sound pressure level (SPL) linked to aircraft overflight affected the heart rate during sleep of residents near airports. Further studies on a larger number of participants over several nights are needed to confirm these results. A larger study (Zijlema et al., 2016) was conducted on the effect of road traffic noise on heart rate (88,336 participants). This study suggests that road traffic noise may be related to increased resting heart rate, supporting that road traffic noise is a risk factor for cardiovascular disease.

Antihypertensive medication use (Thacher et al., 2020) was evaluated in a Danish study of 57,053 participants over a 14-year time period. No association between road traffic noise and filled prescriptions for antihypertensive medications was identified.

Blood pressure effects were evaluated in two studies (Zijlema et al., 2016; Zur Nieden et al., 2016a). The larger study (Zijlema et al., 2016) found no evidence of a relationship between road traffic noise and blood pressure. The smaller study (Zur Nieden et al., 2016a) found no statistical significance in relation to traffic noise on blood pressure.

Blood pressure effects in children – one study (Badihian et al., 2020) evaluated the association between noise annoyance and psychological distress with blood pressure in children. This was a large study (14,400 students) that concluded diastolic blood pressure (the lower number in a blood pressure reading – the pressure in the arteries when the heart rests between beats) and mean atrial blood pressure (average pressure in the arteries during one cardiac cycle) was positively correlated with noise annoyance, and participants with higher psychological distress were 15% more likely to experience higher blood pressure. However, another study (Enoksson Wallas et al., 2019) evaluated traffic noise and hypertension in adolescents and found no conclusive associations between pre- and postnatal noise exposure and blood pressure or hypertension in adolescents.

The above studies have not identified any other health indicators relevant to the assessment of traffic noise on the cardiovascular system that are either significant or sufficient data is available to quantify effects in populations.

3.4.4.1 Meta-analysis studies

Of the additional studies identified in this review, six involved meta-analysis of pooled data from studies published to dates that extend beyond those evaluated by the WHO (van Kempen et al., 2018). These meta-analyses relate to studies on hypertension, IHD, stroke, and myocardial infarction. The following provides an overview of these studies, while Figure 3.4 presents a comparison of the exposure-response relationships identified for these health outcomes from the WHO review and the updated meta-analysis studies. The figure also includes the few additional individual studies identified in this review.

Hypertension

One meta-analysis study (Chen et al., 2021) evaluated study data available to October 2019 in relation to noise and hypertension, pooling data from 11 cohort studies (five representing community noise derived from transport sources). The studies evaluated included studies evaluated in the WHO (2018) review along with three more recent studies. While an association between noise exposure and hypertension was identified, the assessment undertaken was not specific to transport noise as it combined data from transport and occupational studies.

Another meta-analysis study (Dzhambov & Dimitrova, 2018) more specifically reviewed road traffic noise and hypertension based on data from nine studies published from 2011 to 2017. This analysis includes review of a number of the studies identified in the DEFRA review, as well as this review. The overall number of participants in this analysis was 5,514,555 with a median noise level of 57 dB L_{den}. The exposure-response relationship from this study indicates a lower risk than determined in the WHO (2018) review. The quality of the evidence used was evaluated using the GRADE system, with the overall quality of the meta-analysis outcomes determined to be low.

A meta-analysis by Fu et al. (2017) considered 32 studies (which included seven studies on road noise and six studies on aircraft noise) related to the risk of hypertension, published to December 2016. This included studies evaluated in the WHO review (2018) and determined that noise exposure is significantly associated with an increased risk of hypertension. A positive, statistically significant exposure–response relationship is found between the exposure level of noise and the hypertension risk. The review determined exposure-response relationships based on the pooled studies, as well as specific noise sources.

IHD

In relation to IHD, a meta-analysis (Vienneau et al., 2019) was conducted based on studies published to February 2019. The assessment followed the approach adopted by the WHO, including application of the GRADE system for reviewing the risk of bias and overall quality of evidence. Pooled exposure-response functions were determined for exposure to road traffic (13 studies), aircraft traffic (five studies), and rail traffic (three studies) noise as L_{den}. The exposure-response functions were determined to be statistically significant only for road traffic noise. This analysis includes the studies evaluated in the WHO review as well as six new studies (two of which involved large populations), consistent with those identified in the DEFRA review. While the risks are noted to be slightly lower than in the WHO (2018) review, including the additional studies in the meta-analysis has reduced the confidence intervals, particularly for road traffic noise. The exposure-response functions were approximately linear over the whole exposure range, with lowest L_{den} levels typically around 35–45 dBA.

Stroke

In relation to aircraft noise and the incidence of stroke, a meta-analysis (Weihofen et al., 2019) was undertaken based on data published to August 2017. The review identified nine studies of which seven were suitable for inclusion in the meta-analysis, noting the studies were determined to be poor to medium quality. The meta-analysis indicates that aircraft noise increases the risk of stroke, even if the overall finding just fails to reach statistical significance.

Myocardial infarction

The association between road traffic noise and myocardial infarction (MI) has been evaluated using a systematic review and meta-analysis (Khosravipour & Khanlari, 2020). This review identified 13 studies for detailed review and meta-analysis. The findings of these studies indicated inconsistent outcomes, with the statistically significant associations from case-control and cross-sectional studies but not from cohort studies. While meta-analysis resulted in the determination of an exposure-response relationship, it was not considered to be statistically significant. Overall, an association between road traffic noise and MI was determined to be inconclusive.

3.4.4.2 Other studies

It is noted that all the additional studies identified by DEFRA and most of the additional studies identified in this review have been considered in the meta-analysis evaluations discussed above. Four additional studies have been identified in this review that have not been included in the above analysis (Andersson et al., 2020; Recio et al., 2017; Saucy et al., 2020; Shin et al., 2020).

The study conducted by Saucy et al. (2020) was a time-stratified case-crossover study that involved the assessment of aircraft noise and mortality from cardiovascular disease in 24,886 participants near Zurich Airport between 2000 and 2015. The study focused on night-time noise exposures in the 2 hours preceding death and found associations between aircraft noise and mortality for IHD, MI, HF, and arrhythmia (heart rhythm problems). While the outcomes of the study are not comparable to other key studies, due to the limited/specific time period of exposure, the study provides some important observations (Saucy et al., 2020):

- Risk of mortality is higher for female participants than male participants, where it is suggested that females may be more susceptible to stress response, with higher levels of salivary cortisol in response to noise exposure.
- The association between aircraft noise and night-time cardiovascular death was significantly stronger for people living in quiet areas compared with areas with higher night-time levels of road and railway noise and for people living in older buildings, most likely with less sound insulation.
- The study data suggests a threshold for effects in the range of 30 to 50 dB (for 2-hour L_{Aeq})

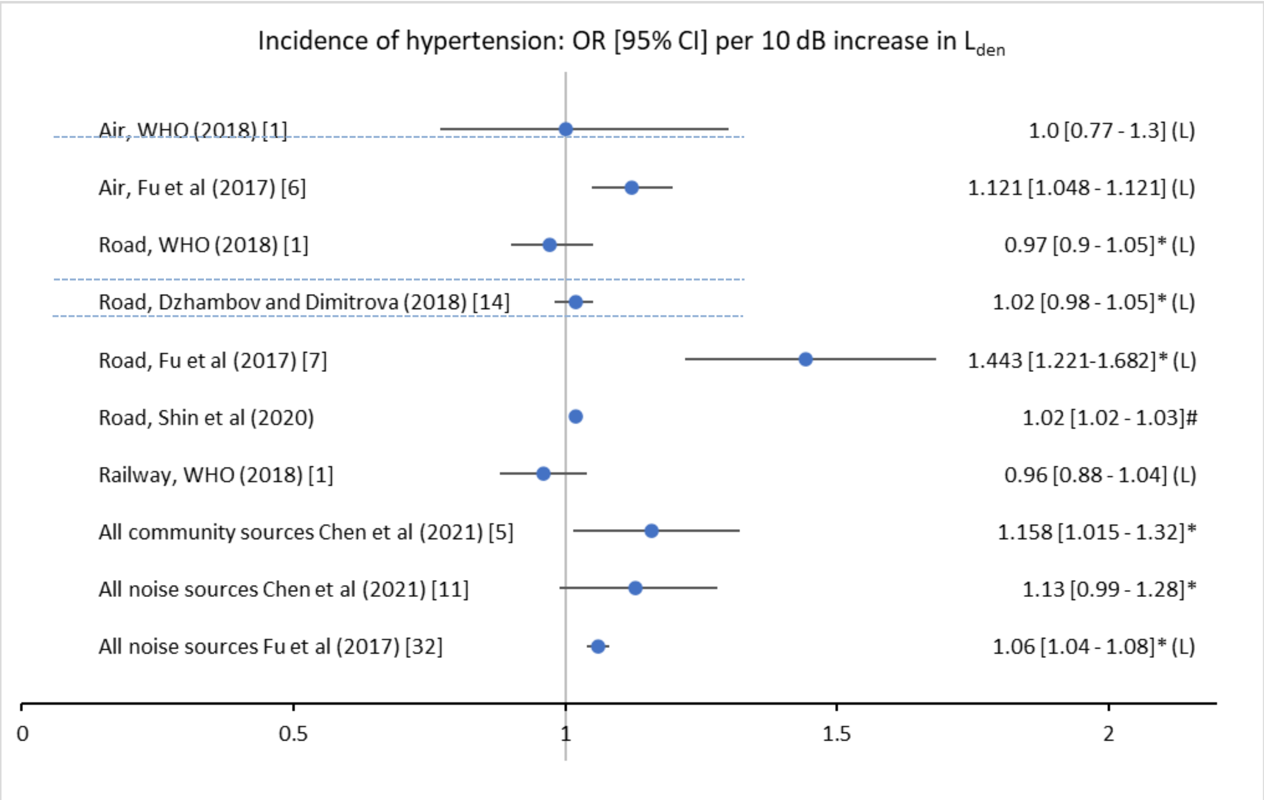
One study (Recio et al., 2017) has been identified that evaluates the impact of transport noise on population mortality, focusing on mortality from specific diseases including respiratory disease, IHD, MI, cerebrovascular disease, pneumonia, chronic obstructive pulmonary disease (COPD), asthma, and diabetes. The study was undertaken on the population of Madrid from 2003 to 2009, with noise levels modelled. The study indicates that it was adjusted for exposure to fine particulate matter ($PM_{2.5}$) and nitrogen dioxide (NO_2). There is limited information presented in the study, however it suggests an association between road traffic noise in urban environments and cardiovascular mortality.

The other two studies have been further reviewed using the GRADE system to determine the potential for bias in each of these studies, as summarised in Appendix C.

The study by Andersson et al. (2020) evaluated road traffic noise, air pollution, and cardiovascular effects in Swedish men over a period of 40 years. The study evaluated incidence of IHD and stroke as well as mortality (all causes and cardiovascular disease). Due to the significant limitation of the study population (male only) the risk of bias was considered to be high. The outcomes of this study are included in Figure 3.4 which shows more significant variability in IHD and stroke incidence than present in the meta-analysis.

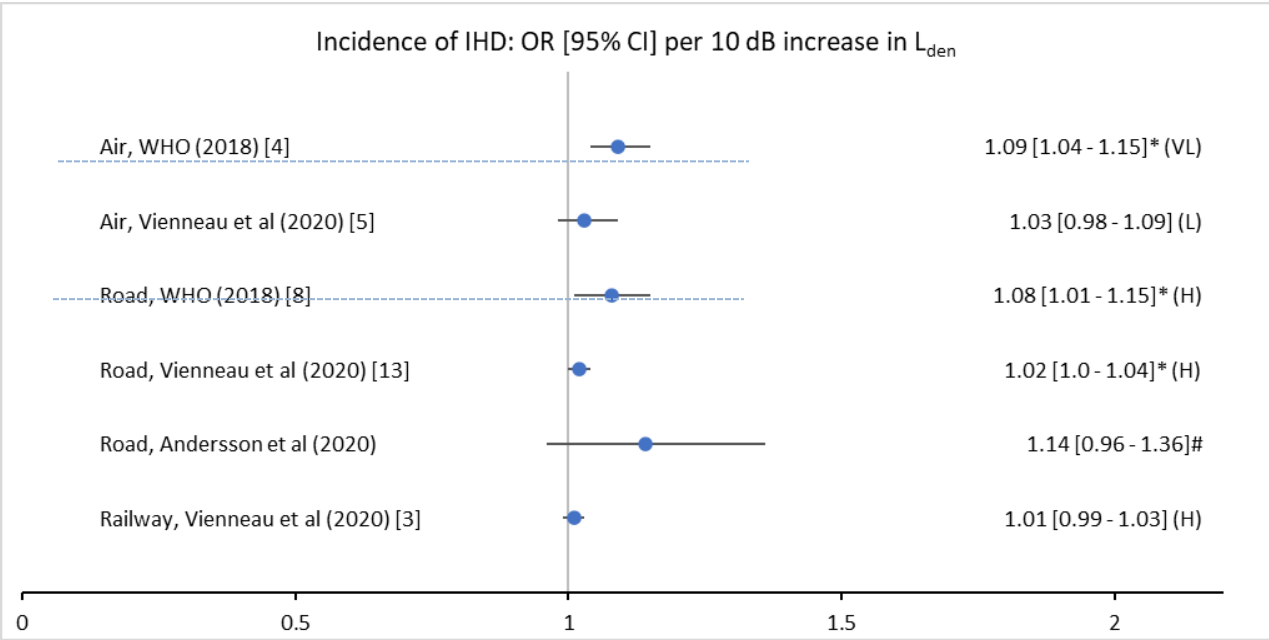
The study by Shin et al. (2020) evaluated long-term exposure to road traffic noise and the incidence of hypertension over a 15-year period (2000 to 2015). The study found a positive association with road traffic noise with exposure-response relationships calculated. The study also notes stronger associations in individuals exposed to lower concentrations of ultra-fine particulates (UFP) and NO_2. The study was evaluated, and the risk of bias was considered to be low. The outcomes of this study are included in Figure 3.4 which shows an odds ratio (OR) (with little variability) in the range identified in the WHO review and similar to the meta-analysis by Dzhambov and Dimitrova (2018).

Figure 3.4 Forest plots for incidence of cardiovascular disease indicators of hypertension (A), IHD (B) and stroke (C) – pooled odds ratios (OR) from meta-analysis per increase in noise related L_{den} by 10 dB, with additional individual studies included (where relevant)



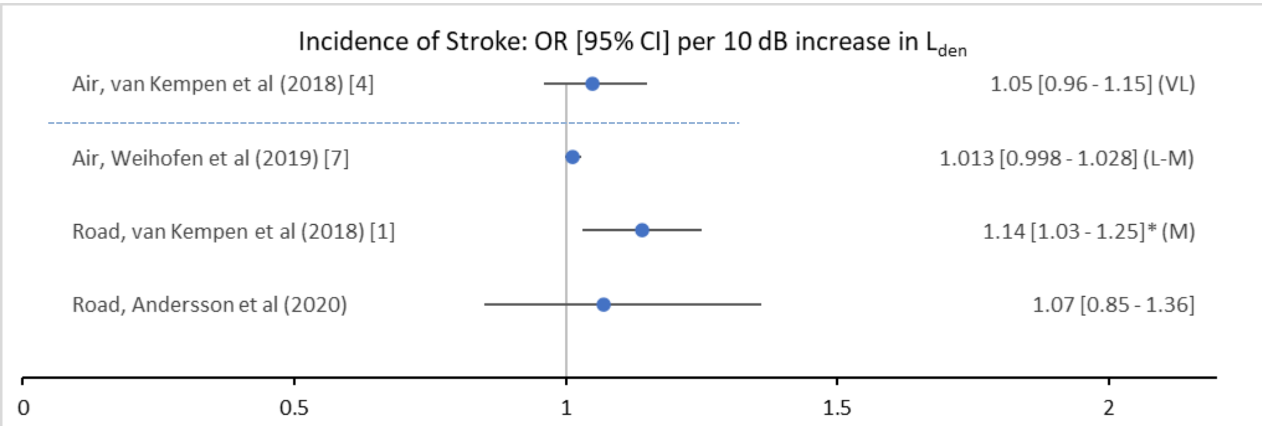
A: Hypertension (incidence) from meta-analysis
Transport mode and reference [n]
OD/RR presented along with quality of evidence ratings: VL = very low, L = low, M = moderate and H = high

* = statistically significant relationships
# = additional individual study (not included in meta-analysis) conducted in Canada (Toronto) with 701,174 participants over 15 years (Shin et al., 2020) (high quality study)



B: IHD (incidence) from meta-analysis
Transport mode and reference [n]
OD/RR presented along with quality of evidence ratings: VL = very low, L = low, M = moderate and H = high

* = statistically significant relationships
# = additional individual study (not included in meta-analysis) conducted in Sweden with 6304 male participants over 40 years (Andersson et al., 2020) (low quality study)



C: Stroke (incidence) from meta-analysis
Transport mode and reference [n]
OD/RR presented along with quality of evidence ratings: VL = very low, L = low, M = moderate and H = high

* = statistically significant relationships
# = additional individual study (not included in meta-analysis) conducted in Sweden with 6304 male participants over 40 years (Andersson et al., 2020) (low quality study)

A recent ‘Letter to the editor’ (Kawada, 2021) provides further commentary on cardiovascular mortality risks based on the available studies and meta-analysis. In particular, the letter notes the identification of an association between road traffic noise and IHD mortality, however no significant association was identified for aircraft noise and IHD mortality. This was partly due to the limited number of studies available, however CVD mortality for aircraft noise is changed by intermittency and hence it is important to also evaluate the characteristics of noise and aspects such as habituation to noise. This means that while an association has not been identified, it does not mean that aircraft noise does not affect CVD mortality. The noise measures used for analysis, however, need to be further considered to ensure these are appropriate for the type of noise source.

3.4.5 Overall evaluation and recommendations

It is clear from the available information/data that there is sufficient evidence of a causal relationship between exposure to environmental noise and CVD outcomes (enHealth, 2018; WHO, 2018). The health measures to be used in the quantification of cardiovascular effects is important.

The WHO (2018) review included hypertension regardless of the low quality of evidence and inconsistent outcomes, as it was considered to be a key health indicator. It is expected that effects captured in the measure of hypertension would be captured in the IHD outcomes, therefore the need to include hypertension as a health indicator regardless of quality is questionable. It is also important to consider how the health measure would be quantified, as the incidence of hypertension in the community is not well reported. The more recent evaluations of hypertension outcomes have not changed the quality of evidence in relation to transport sources (remaining low). The only exposure-response relationships with statistically significant outcomes relates to road traffic noise, where the data does not suggest the WHO (van Kempen et al., 2018) evaluation should change. There is insufficient quality and consistency of evidence to include evaluations of hypertension for rail and air traffic noise sources. If hypertension needs to be evaluated, then it is recommended that the exposure-response relationship established from all sources (Fu et al., 2017) be adopted for all transport noise sources.

In relation to IHD, the meta-analysis by Vienneau et al. (2020) is considered to be an update of the WHO meta-analysis, incorporating all published studies to 2019. Vienneau et al. (2020) identified lower risks, but a slightly higher quality of evidence for aircraft and railway noise, and a narrower 95% CI for road traffic noise. For the assessment of noise impacts in urban areas, it is recommended that these more recent exposure-response relationships are adopted. It is noted that a number of studies have observed a steeper slope of the exposure-response function for individuals in areas with quiet background noise (similar to observations on noise annoyance). Hence, for the assessment of IHD impacts from road noise sources in rural areas (or areas with low background noise levels) the more conservative exposure-response function from the WHO review (2018) may be considered, which is considered relevant for the New Zealand context – refer to section 3.2.4. In relation to the assessment of aircraft noise, the WHO (van Kempen et al., 2018) relationship is recommended, as the relationship from Vienneau et al. (2020) was not statistically significant.

Table 3.2 presents a summary of the exposure-response relationships recommended for use in this assessment.

Note that in the cost model, the exposure-response function for road noise in rural areas only in order to take a worst-case conservative approach. This is explained further in section 5.2.3.3.

Table 3.2 Recommended exposure-response functions for quantification of cardiovascular disease

Health outcome	Noise metric	Lowest level of exposure (dB)	Exposure-response relationship per 10 dB increase (RR = relative risk or OR = odds ratio) [95% confidence interval]	Quality of evidence
Road noise – incidence of IDH – urban areas	L_{den}	53	RR = 1.02 [1.0–1.04]	High (Vienneau et al., 2019)
Road noise – incidence of IDH – rural areas	L_{den}	53	RR = 1.08 [1.01–1.15]	High (van Kempen et al., 2018)
Railway noise – incidence of IHD	L_{den}	N/A	RR = 1.01 [0.99–1.03]	High (van Kempen et al., 2018)
Aircraft noise – incidence of IHD	L_{den}	47	RR = 1.09 [1.04–1.15]	Very low (van Kempen et al., 2018)

3.5 Metabolic outcomes

3.5.1 Overview of adverse effect

Consistent with the review by Münzel et al. (2020) that provided a more detailed review of the mechanisms for adverse effects on the cardiovascular system, these mechanisms are also considered to result in metabolic effects. The hypothesis is that noise exposure is related to stress hormone-mediated increase in cortisol and deposition of fat centrally, as well as other impacts on metabolic functioning and/or adverse effects of disturbed sleep on metabolic and endocrine function (also refer to section 3.4.1) (Kim et al., 2017; Münzel et al., 2020; Sparrow et al., 2020). This hypothesis is somewhat supported by a limited number of studies that have found associations between noise exposure and diabetes.

3.5.2 WHO evaluation

WHO (2018) identified metabolic outcomes (based on L_{den}), specifically the prevalence, incidence, hospital admissions, or mortality due to type 2 diabetes and obesity, to be important health outcomes for the assessment of exposure to environmental noise.

The available studies in relation to metabolic effects were evaluated in detail by van Kempen et al. (2018). This review identified eight studies covering the period 2000 to August 2015 for the assessment of metabolic effects.

In relation to diabetes, the number of studies available was limited, and the studies were found to be mixed (in terms of an association), imprecise, with the quality of evidence for road, rail, and aircraft noise rated as very low.

Similarly for the assessment of obesity, a statistical non-significant association was found between road traffic noise, BMI, and waist circumference. For aircraft and railway noise, statistically significant associations were found with waist circumference, but not BMI (with BMI changes for aircraft noise also not considered clinically significant). The results were considered inconsistent, and the quality of evidence rated as very low.

On this basis, no quantitative exposure-response relationships were recommended for the assessment of these effects. A better quality of evidence was required to enable these effects to be quantified.

3.5.3 DEFRA evaluation

The review completed by DEFRA (van Kamp et al., 2020; van Kamp et al., 2019) identified an additional eight studies on the metabolic system for the period 2015 to March 2019. These studies related to the incidence of diabetes, change in BMI, change in waist circumference, incidence of obesity, and incidence of overweight. All these additional studies were identified in the literature review conducted for this assessment.

These additional studies add to the studies identified in the WHO (2018) review (van Kempen et al., 2018). The DEFRA review suggested that the evaluation of diabetes in relation to road and air traffic noise should be revised within the new studies. However, the review notes that despite additional studies being available, the overall number of studies remains limited with exposure-response relationships not expected to be robust. Similar outcomes relate to the assessment of obesity.

The DEFRA review did not undertake any meta-analysis of the studies, and the studies identified were not further evaluated using the GRADE system. In relation to the assessment of effects from aircraft noise, the review conducted by NatCen (Grollman et al., 2020) provided a review of the strength of evidence (using the GRADE system) in relation to more recent studies on metabolic effects, concluding that the available evidence remains low or very low quality.

3.5.4 Additional studies

The literature review conducted for this assessment identified 21 studies published in the period 2015 to March 2021. This includes the eight that were identified in the DEFRA review.

Of the additional studies identified in this review, three were further excluded on the basis that they were duplicates and did not provide new study data, were evaluated on the basis of noise annoyance (or other parameters) not noise exposure levels, or there were no exposure-response relationships evaluated.

It is noted that a number of the studies identified relate to the assessment of co-exposures to noise and air pollution, both of which have been associated with metabolic disease. These co-exposures make interpretation of the outcomes of many of these studies complex.

3.5.4.1 Diabetes

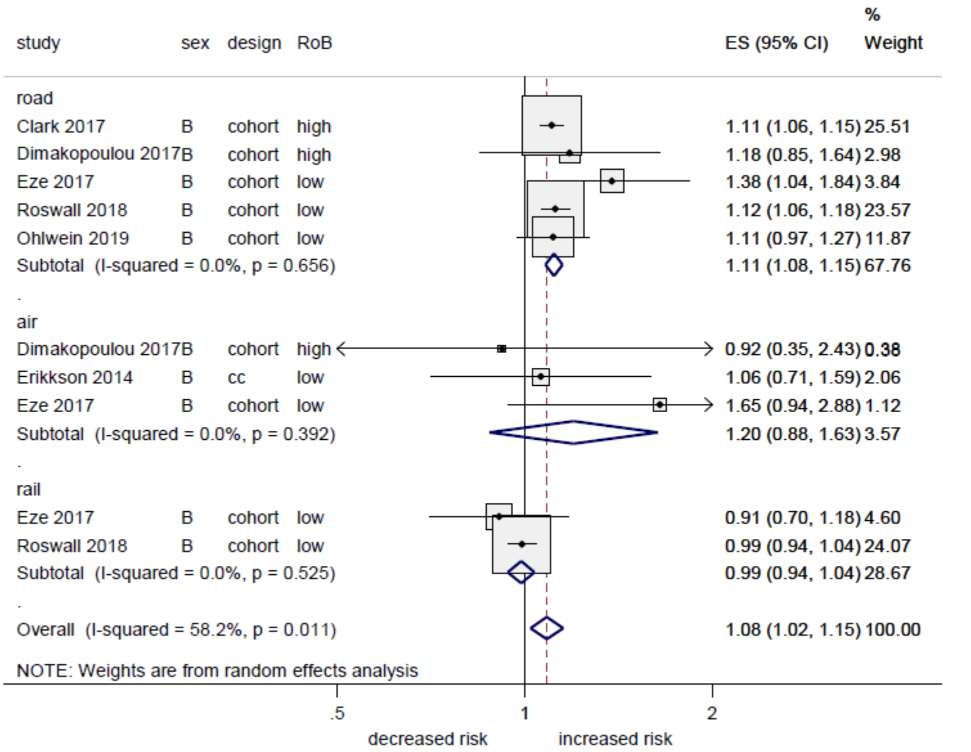
Three of the studies identified are more recent meta-analysis studies on diabetes (Vienneau et al., 2019; Wang et al., 2020; Zare Sakhvidi et al., 2018).

The meta-analysis by Wang et al (Wang et al., 2020) considered studies published between 2009 and 2019 in relation to a range of noise sources (transport, residential, and occupational). The paper identified eight studies (five cohort and three cross-sectional) and the risk of diabetes. The quality of the studies was evaluated using the Newcastle-Ottawa Scale (NOS). Overall, the potential for publication bias in these studies was determined to be low. Pooled exposure-response relationships were presented, with the odds ratio determined to be more conservative than that determined by Vienneau et al. (2019), mainly due to the inclusion of studies relating to residential (general) and occupational noise sources. The transport studies included in this meta-analysis were included in the more focused meta-analysis by Vienneau et al. (2019).

The meta-analysis conducted by Zare Sakhvidi et al. (2018) was based on published studies to September 2017. The quality of the studies was evaluated using the NOS. The review identified six cohort, six cross-sectional, and three case-control studies relating to the incidence of diabetes and transport noise as well as occupational noise sources. This resulted in 474,474 participants and 18,441 diabetes mellitus cases. Six of the studies were determined to be of high quality, with nine considered to be of low to fair quality. Pooled exposure-response relationships were presented for the incidence of diabetes for a 5 dB increase in noise as L_{den}. Given the limited number of studies and the variability observed in these studies, it was difficult to compare associations between different noise sources. However, the strongest association was identified for aircraft noise, followed by road traffic noise. No association was found in relation to railway noise.

The meta-analysis conducted by Vienneau et al. (2019) was based on studies published to February 2019 (including key studies incorporated in the meta-analysis by Zare Sakhvidi et al. (2018)). The assessment followed the approach adopted by the WHO, including application of the GRADE system for reviewing the risk of bias and overall quality of evidence. Pooled exposure-response functions were determined for exposure to road traffic (five cohort studies), aircraft traffic (two cohort and one case-control study) and rail traffic (two cohort studies) noise as L_{den}. Figure 3.5 presents the relative risks from all studies evaluated, and the pooled outcomes from the meta-analysis. The exposure-response functions were determined to be statistically significant only for road traffic noise. The overall risk of bias for these studies was determined to be low. The evaluation determined that transportation noise is an important risk factor for diabetes.

Figure 3.5 Forest plot of updated meta-analysis for the incidence of diabetes and transport noise (per 10 dB L_{den}) by source (Vienneau et al., 2019) (note the only relationship with statistical significance is for road traffic noise)



In relation to the assessment of aircraft noise, a number of individual studies and the meta-analysis from Vienneau et al. (2019) was included in the review of aircraft noise by NatCen (Grollman et al., 2020). This included GRADE analysis of the available and relevant studies. The Vienneau et al. (2019) analysis was determined to have a low potential for bias. The overall GRADE evaluation for the quality of evidence on relation to the incidence of diabetes from aircraft noise sources was determined to be low quality.

Another three studies were also identified that evaluated effects of exposure to road traffic noise and diabetes, two of which relate to the incidence of type 2 diabetes (Jørgensen et al., 2019; Shin et al., 2020) and one that relates to gestational diabetes (Pedersen et al., 2017). Appendix C presents a review of these additional studies using the GRADE system. The risk of bias for these studies was determined to be low to moderate. The studies indicated mixed outcomes. There was no evidence of an association between road traffic noise and incidence of gestational diabetes. For the incidence of type 2 diabetes, one study showed no association (with suggestive evidence of an association for exposures in urban areas only). However, another study involving a larger, more representative population (Shin et al., 2020) showed an association with an exposure-response relationship (RR = 1.08 [1.07–1.09]). The association identified sits with the range of associations reported in other studies relating to road traffic noise.

The additional studies identified do not add to or change the quality of evidence as evaluated in the meta-analysis conducted by the WHO (van Kempen et al., 2018) or Vienneau et al. (2019).

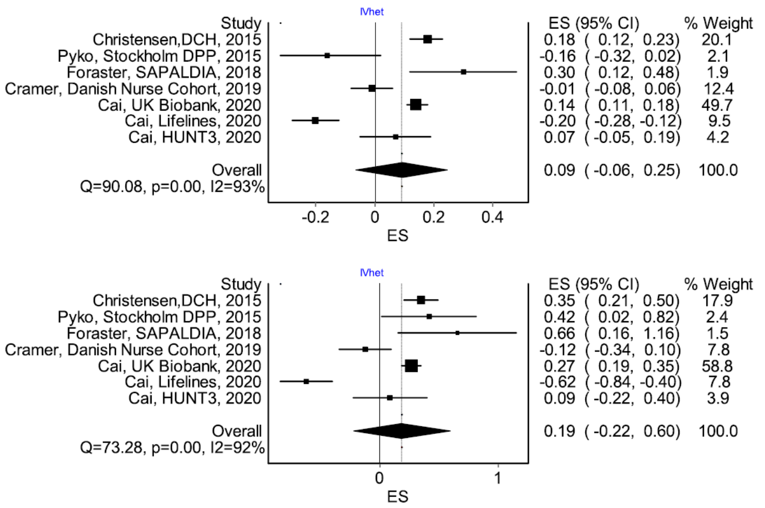
3.5.4.2 Obesity

Three of the additional studies identified related to meta-analysis of studies on the effects of noise on obesity, measured on the basis of waist circumference, and BMI (An et al., 2018; Cai et al., 2020), with one of these studies relating to childhood obesity (Wang et al., 2020).

The study by An et al. (2018) was based on published studies to February 2018, including studies evaluated by the WHO. Eleven studies were identified that showed a positive association between chronic noise exposure and obesity indicators. This included from six studies related to waist circumference and seven studies related to BMI. The quality of the studies identified was evaluated using the GRADE system. In relation to waist circumference, an association was identified for exposures above 55–60 dB as L_{den} with meta-analysis used to determine a pooled exposure-response relationship for all studies combined, with no distinction between noise sources. No association was found between noise exposure and BMI. The overall quality of evidence in relation to obesity outcomes was determined to be low to very low.

The study by Cai et al. (2020) reports the results of a study on three population cohorts in Europe between 2006 and 2013 in the UK, the Netherlands, and Norway for road traffic noise and obesity markers, BMI, waist circumference, and odds of obesity. The study identified exposure-response relationship for those exposed to L_{den} > 55 dB. The study, while it considered a large pooled population of 504,271 participants, also included meta-analysis of BMI and waist circumference indicators to update the evaluation undertaken by the WHO. The analysis did not include an assessment of study quality. The meta-analysis of pooled data determined outcomes slightly higher than those reported in the 2018 WHO evaluation, and while the additional data has strengthened the evidence base in terms of numbers of studies, individuals, and range of noise exposure (42–89 dB), the analysis showed more uncertainty (ie, wider 95% CI). Review of the relationships for BMI and waist circumference (included in Figure 3.6) shows significant variability in outcomes.

Figure 3.6 Forest plot of updated meta-analysis for the incidence of BMI (top plot) and waist circumference (lower plot) and road traffic noise (per 10 dB L_{den}) (Cai et al., 2020) (ES = effect size)



The study by Wang et al. (2020) evaluated the available evidence on noise exposure and childhood obesity, and concluded that the evidence only supports a weak association, with outcomes largely inconclusive.

Another four studies were also identified that evaluated effects of exposure to road traffic noise and obesity indicators (with one study also evaluating rail and air traffic noise), including BMI and waist circumference (Cai et al., 2020; Cramer et al., 2019; Foraster et al., 2018; Weyde et al., 2018), three of which are included in the updated meta-analysis conducted by Cai et al. (2020). Appendix C presents a review of these additional studies using the GRADE system. The risk of bias for these studies was determined to be low to high. For road traffic noise, the studies show mixed outcomes in relation to BMI and waist circumference (or obesity). No associations were identified for air traffic noise (one additional study) and only mixed outcomes were reported for rail noise (one study).

The study by Weyde et al. (2018) relates to BMI in children where road noise exposures occurred during pregnancy or childhood. The only association identified related to road traffic noise exposure during pregnancy and BMI trajectory in children, with mothers exposed to higher levels of road traffic noise during pregnancy having children with a lower BMI at birth but higher BMI at 8 years of age. The association, however, cannot be excluded as being a result of chance. No other associations were identified.

The additional studies identified do not add to or change the quality of evidence as evaluated in the meta-analysis conducted by the WHO (van Kempen et al., 2018) or by Cai et al. (2020) for road traffic noise. Insufficient new information is available to indicate that associations between rail and air traffic noise and obesity can be reliably established.

3.5.5 Overall evaluation and recommendations

Overall, the quality of evidence for quantifying effects of transport noise on metabolic outcomes (diabetes and obesity) is low and these measures should not be included in any quantitative evaluation of population health burden.

The only statistically significant relationship identified relates to exposure to road noise and the incidence of diabetes, where the relationship established in the more recent meta-analysis by Vienneau et al. (2019) of RR = 1.11 [1.08–1.15] per 10 dB L_{den} may be considered for inclusion in this assessment.

3.6 Cognitive impairment (children)

3.6.1 Overview of adverse effect

There is evidence for effects of noise on cognitive performance in children, in particular lower reading performance (WHO, 2011, 2018). Noise in classrooms affects children in many ways, including lowering their motivation, reducing speech intelligibility, lowering listening comprehension and concentration, producing annoyance and disturbance, and increasing restlessness. As a result, children exposed to noise at school may experience poorer reading ability, memory, and performance. Cognitive impairment could also be linked to noise exposure at home during night-time hours, which can cause low mood, fatigue, and impaired task performance the next day. Noise at home may also be linked to hyperactivity and inattention problems, which can cause lower academic performance (EEA, 2020).

3.6.2 WHO evaluation

The earlier WHO evaluation (WHO, 2011) focused on evidence from a major study in the EU (RANCH). The study found an exposure-response relationship between noise and cognitive performance in children for aircraft noise, but the relationship between performance and noise for road traffic was much less clear (Stansfeld et al., 2005a; Stansfeld et al., 2005b; WHO, 2011, 2018). WHO (2011) recommended the use of the aircraft noise relationships to assess the impact of noise on children's cognitive performance.

The WHO (2018) (Clark et al., 2018a) review identified cognitive impairment as a critical health outcome for the assessment of exposure to noise, particularly for the most sensitive group – children, with the key health measures being reading and oral comprehension. Other measures include short and long-term memory deficit, attention deficit, and executive function deficit. The WHO (2018) evaluation considered that a delay of one month is of relevance for assessing absolute risks of exposure.

The review (Clark et al., 2018a) identified 34 papers from 2000 to June 2015.

For the assessment of aircraft noise, 14 studies were identified that related reading and oral comprehension, where the quality of evidence was rated as moderate. Most of the studies showed a statistically significant association between higher aircraft noise and poorer reading comprehension. The relationship was supported by other evidence relating to cognition (including standardised test performance and poorer long-term memory). There was no substantial evidence of an association with attention or executive function. For the assessment of aircraft noise, a 1 to 2-month delay per 5 dB increase in L_{den} was recommended.

For the assessment of road traffic noise, two cross-sectional studies reported effects of noise exposure on children's cognition (Clark et al., 2018a), which included the RANCH study. This did not find an association for exposures to children in the range 31–71 dB as $L_{Aeq,16\ hr}$. The quality of evidence was rated as very low. Low and very low-quality evidence was identified for the effect of noise exposure on cognitive impairment using standardised tests, long-term memory (with no studies on short-term memory), attention, and executive function. No exposure-response function was identified and no quantitative assessment of impacts on cognitive function was recommended.

There was a lack of studies available for the assessment of rail traffic noise on cognitive outcomes. The few studies that were available were considered to provide very low-quality evidence of an association, with cognition measured with standardised tests and long-term memory. No association was identified for attention in children. No exposure-response function was identified and no quantitative assessment of impacts on cognitive function was recommended.

3.6.3 DEFRA evaluation

DEFRA (ARUP, 2020; Clark et al., 2020) conducted a review of studies published after the WHO review, between mid-2015 and March 2019. In relation to cognitive effects, nine studies were identified for systematic review using the GRADE system. These nine studies included three studies identified in this review and an additional six studies. Two of the studies related to exposures by older adults, with seven related to exposures by children. The risk of bias for these studies was predominantly low however four studies had a high risk of bias, or the risk of bias could not be determined. Overall, the evidence for an effect of environmental noise on cognition for children and adults was mixed. For children, the NORAH study (Klatte et al., 2017; Spilski et al., 2016) found results similar to the earlier RANCH study, where an association between aircraft noise and reading comprehension was identified. The DEFRA evaluation determined the following in relation to the quality of evidence:

- Reading comprehension – the quality of evidence was very low for air and road traffic sources (no studies for rail).
- Mathematics – the quality of evidence was very low for road traffic sources (no studies for air or rail).
- Student distraction – the quality of evidence was very low for aircraft sources (no studies for road or rail).
- Adult cognition – the quality of evidence was very low for road traffic sources (no studies for air or rail).
- Working memory and attention in children – the quality of evidence was low for road traffic sources (no studies for air and rail).

DEFRA conducted a review of the new studies with the WHO (2018) findings. The WHO review identified a moderate quality of evidence for an effect of aircraft noise on children’s reading and oral comprehension and a low quality of evidence for no substantial effect of road traffic noise on children’s reading and oral comprehension (section 3.6.2). The studies reviewed by DEFRA indicated a low quality of evidence for an effect of aircraft and road traffic noise on children’s reading comprehension. This was considered by DEFRA to potentially be a reflection of the low number of studies reviewed by them.

Overall, the DEFRA evaluation concluded that the outcomes of the WHO (2018) evaluation remain unchanged for comprehension. An update of the assessment of road traffic noise on executive function/working memory and attention is suggested.

3.6.4 Additional studies

In relation to the assessment of cognitive effects, this study identified seven studies published between 2015 and March 2021. This includes three studies incorporated and reviewed (using the GRADE system) by DEFRA.

One study was excluded from systematic review (Huang et al., 2021) as this focused on existing experimental and epidemiological studies related to the underlying mechanisms that relate to chronic noise exposure and cognitive impairment and degenerative dementia. The paper indicates that several hypotheses have been proposed regarding the mechanisms of noise exposure, cognitive dysfunction, and dementia, although the evidence is insufficient to draw a conclusion. These hypotheses include the following (Huang et al., 2021):

- Whether it is the noise pressure or peripheral hearing loss induced by noise that leads to neuronal damages is not clear. Current evidence suggests that the two may coexist.
- The way in which abnormalities such as neuroinflammation, psychological stress, changes in redox state, and excitotoxicity after noise exposure cause neuropathological changes in the hippocampus (as triggers or persistent exacerbation) has yet to be determined. That some early changes seem to be able to recover in chronic exposure models while cognitive dysfunction and neuropathological changes are often continuous and non-recoverable indicates that further mechanism research is warranted.

- The uncertainty identified by Huang et al. (2021) appears to be replicated in the available studies, where mixed and inconclusive outcomes are reported particularly for road and rail transport noise sources. This may be due to the limited number of high-quality studies that address this health endpoint.

Three additional studies (not evaluated by DEFRA) (Robinson et al., 2021; Tzivian et al., 2020; Weuve et al., 2020) have been further reviewed using the GRADE system. None of these additional studies change the outcomes or quality of evidence in relation to transport noise and cognitive effects in children.

One of the studies related to children and the effect of noise in classrooms (including aircraft noise) on student concentration (Robinson et al., 2021). While the study identified an association between noise intensity and student learning disorders, the risk of bias for the study was determined to be high.

The other two studies (Tzivian et al., 2020; Weuve et al., 2020) focused on the effects of community noise (ie, noise from all sources) on older adults and mild cognitive impairment (MCI), cognitive performance, cognitive decline, and Alzheimer's disease (AD). The risk of bias relevant to these studies ranges from low to moderate. Both studies identified an association between increased noise exposure and decreased cognitive function (as MCI and cognitive performance) and increased risk of AD. Associations were found to be stronger where indoor noise was considered in the evaluation, and for individuals with severe depressive symptoms.

3.6.5 Overall evaluation and recommendations

Overall, the quality of evidence for exposure to transport noise to impact on cognitive outcomes remains limited. The evaluation presented by the WHO (2018) remains unchanged with consideration of additional studies. For aircraft traffic noise there is a moderate quality of evidence for effects on reading comprehension for children, where an exposure-response relationship is recommended for use in quantitative assessments, as follows:

Table 3.3 Recommended exposure-response relationships for assessing cognitive impairment

Health outcome	Noise metric	Lowest level of exposure (dB)	Exposure-response relationship per 10 dB increase (RR = relative risk or OR = odds ratio) [95% confidence interval]	Quality of evidence
Aircraft noise – cognitive impairment in children (as reading and oral comprehension)	L_{den}	55	1 to 2-month delay per 5 dB increase	Moderate

Insufficient evidence is available to assess cognitive impairment for other noise sources, or other cognitive outcomes.

3.7 Sleep disturbance

3.7.1 Overview of adverse effect

Sleep serves to facilitate vital functions in our body. It is relatively well established that night-time noise exposure can have an impact on sleep (enHealth, 2018; WHO, 2009, 2011). Noise can cause difficulty in falling asleep, awakening, and alterations to the depth of sleep, especially a reduction in the proportion of healthy rapid eye movement sleep. Other primary physiological effects induced by noise during sleep can include changes in glucose metabolism and appetite regulation, impaired memory consolidation, and a dysfunction in blood vessels. Long-term sleep disturbance can also lead to cardiovascular health issues (WHO, 2011, 2018). Exposure to night-time noise also may induce secondary effects, or so-called aftereffects. These are effects that can be measured the day following exposure, while the individual is awake, and include increased fatigue, depression, and reduced performance.

3.7.2 WHO evaluation

Studies are available that have evaluated awakening by noise, increased mortality (ie, increase in body movements during sleep), self-reported chronic sleep disturbances, and medication use (EC, 2004). The most easily measurable outcome indicator is self-reported sleep disturbance, where there are a number of epidemiological studies available. From these studies, the WHO (2009, 2011, 2018) identified an exposure-response relationship that relates to the percentage of persons sleep disturbed (%SD) and highly sleep disturbed (%HSD) to total levels of noise reported as L_{night} (ie, average noise levels during the night, which is an 8-hour time period, as measured outdoors). The relationship adopted relates to the assessment of road traffic noise, with other relationships for aircraft and rail traffic noise. The review by the WHO (2018) concluded that the key outcome of %HSD was considered most appropriate for determining actions and outcomes in relation to transport noise. Hence this assessment has focused on %HSD.

The WHO review (Basner & McGuire, 2018) identified 74 studies for the period 2000 to 2015 of which 33 cross-sectional studies were used in meta-analysis. The studies related to specific sources with eight studies relating to aircraft noise, 15 studies relating to road traffic noise, and six studies relating to rail noise. The studies related to road traffic sources were judged to provide moderate quality of evidence in relation to cortical awakenings and self-reported sleep disturbance. For the other noise sources, the quality was evaluated as ‘very low’ for all investigated sleep outcomes.

Overall, it was concluded that transportation noise affects objectively measured sleep physiology and subjectively assessed sleep disturbance in adults. For other outcome measures and noise sources, the examined evidence was conflicting or only emerging (van Kamp et al., 2020).

3.7.3 DEFRA evaluation

The DEFRA review (van Kamp et al., 2020; van Kamp et al., 2019) identified an additional 42 studies addressing the effects of noise on sleep for the period 2015 to 2019. This included 12 additional studies for aircraft noise, 10 studies for road noise, and six studies for rail noise. The evaluation completed by van Kamp et al. (2020) included a detailed assessment of these additional studies using the GRADE system. For the assessment of aircraft noise, the new studies on sleep disturbance suggested the need for the meta-analysis completed by WHO to be updated. This could also be considered for road and rail noise; however, the new studies did not show any significant differences in outcomes.

The DEFRA review did not undertake any updated meta-analysis, however a number of the studies identified in the DEFRA review were also identified in this review. The DEFRA review also identified an additional eight studies not initially identified in this review (Bodin et al., 2015; Douglas & Murphy, 2016; Holt et al., 2015; Joost et al., 2018; Kim et al., 2019; Martens et al., 2018; Paiva et al., 2019; Pultznerova et al., 2018).

The risk of bias in these studies ranged from low to medium.

3.7.4 Additional studies

The literature review conducted for this assessment identified 26 studies published in the period 2015 to March 2021. These included 14 that were identified in the DEFRA review. The studies evaluated in the DEFRA review have not been individually evaluated again as the GRADE system of review has already been applied to these studies.

Of the additional studies identified in this review, five were further excluded on the basis that they were duplicates and did not provide new study data, assessed population health burdens (using existing exposure-response relationships), were evaluated on the basis of noise annoyance (or other subjective parameters) not noise exposure levels, or there were no exposure-response relationships evaluated.

One of the excluded papers was a study that pooled results from three small laboratory studies with 237 individuals (Elmenhorst et al., 2019). These pooled results showed that the three major transport noise sources differ in their impact on sleep. Results indicate that different traffic noise sources induce different awakening probabilities, even at equal maximum A-weighted SPL, and even after adjusting for acoustical parameters as well as physiological parameters. At equal maximum A-weighted SPL, the awakening probability due to the three traffic noise sources increased in the order aircraft < road < railway noise. These findings support results from field studies conducted by the authors (Basner et al., 2011; Elmenhorst et al., 2010; Elmenhorst et al., 2012; Marks et al., 2008) that also indicated a higher awakening probability due to railway noise in comparison to aircraft noise, as well as outcomes on sleep continuity. The order, however, is inverse to that associated with noise annoyance. Further, the susceptibility to noise-induced awakenings or arousals is highly variable among individuals. The exposure-response functions adopted for assessing noise impacts on communities represent an individual with average noise susceptibility. These relationships do not address susceptible groups.

None of the additional studies identified involved a more up-to-date meta-analysis than present in the WHO review (Basner & McGuire, 2018).

Following this review, seven papers were identified that required further review in relation to quality. These relate to aircraft noise (three studies), road traffic noise (two studies), and combined road and aircraft or road, rail, and aircraft traffic noise (two studies). These have been evaluated following the GRADE system.

Appendix C presents a summary of the seven studies evaluated, along with the risk of bias ranking. The papers were predominantly considered to be of low to moderate quality (moderate to high risk of bias) with one study of high quality (low risk of bias). The studies report a mixed range of sleep measures, with only a few specifically addressing %HSD.

The papers reviewed by DEFRA (van Kamp et al., 2020), as well as the additional seven studies reviewed in this assessment, have been further considered in conjunction with the GRADE reviews completed in the WHO review (Basner & McGuire, 2018). The GRADE system has then been used to review the overall quality of evidence as assessed by Basner et al. (2018). These reviews are included in Appendix C and show that the additional studies do not significantly change the overall quality of evidence in relation to sleep disturbance (as %HSD from self-rated sleep disturbance surveys/measures), however the overall quality of evidence should be considered low to moderate.

3.7.5 Overall evaluation and recommendations

While additional studies are available in relation to the assessment of sleep disturbance from traffic sources, the outcomes from these reviews are mixed. While the DEFRA review suggests the need to update the meta-analysis for aircraft noise, this has not been undertaken and review of the new studies suggests that the WHO evaluation for aircraft noise (Basner & McGuire, 2018) may be more conservative than the outcomes from more recent studies. On this basis, the exposure-response relationships adopted in the WHO (2018) evaluation should be retained for the assessment of sleep disturbance within the community. These relationships are summarised in Table 3.4.

Table 3.4 Recommended exposure-response functions for quantification of self-reported sleep disturbance as %HSD*

Health outcome	Noise metric	Lowest level of exposure (dB)	Exposure-response relationship per 10 dB increase (RR = relative risk or OR = odds ratio) [95% confidence interval] Regression equation	Quality of evidence (all studies to 2021) and reference
Road traffic noise	L_{night}	43	OR = 2.13 [1.82–2.48] %HSD = 19.7885–0.9336 × L_{night} + 0.0126 × L_{night}^2	Moderate (Basner & McGuire, 2018)
Railway noise	L_{night}	33	OR = 3.06 [2.38–3.93] %HSD = 67.5406–3.1852 × L_{night} + 0.0391 × L_{night}^2	Moderate (Basner & McGuire, 2018)
Aircraft noise	L_{night}	35	OR = 1.94 [1.61–2.33] %HSD = 16.7885–0.9293 × L_{night} + 0.0198 × L_{night}^2	Moderate (Basner & McGuire, 2018)

* Relationships identified relate to exposures by adult populations

For sleep disturbance, the relevant risk level that is protective of health has been determined by the WHO (2018) to be 3%HSD.

3.8 Other health effects

3.8.1 General

This section provides a review of other health outcomes from exposure to noise that have been evaluated in the literature, but not found to provide sufficient quality of evidence and/or robust associations for these to be included in quantitative analysis of health (WHO, 2018). A number of these health outcomes have been further reviewed by DEFRA (ARUP, 2020; Clark et al., 2020), where studies published from 2014/2015 to March 2019 have been evaluated. Additional studies identified in this review have also been considered in the discussion below.

3.8.2 Dementia

Dementia was not a specific health outcome evaluated in the studies underpinning the WHO (2018) review. In some studies, dementia has been evaluated alongside some of the studies evaluating noise exposures and cognitive effects in adults (also refer to section 3.6). The DEFRA review included an assessment of studies specifically evaluating dementia and other neurodegenerative diseases (ARUP, 2020; Clark et al., 2020). The DEFRA review identified nine studies (most of which were also identified in this review), two of which were excluded (as they did not measure or assess noise exposures). The studies evaluated in detail, using the GRADE system, included those identified in the literature review conducted for this assessment. This study also identified an additional two studies published to March 2021 which have been reviewed in relation to cognitive effects in adults (section 3.6) (Tzivian et al., 2020; Weuve et al., 2020). One additional study published in September 2021 has also been identified and has been included in this assessment (Cantuaria et al., 2021).

The studies identified considered a range of dementia outcomes including medical diagnoses of Parkinson's disease, dementia (in general) or Alzheimer's disease, hospitalisations for dementia-related illnesses, as well as cognitive tests of dementia or dementia symptoms or precursors to dementia.

Review of the quality of evidence by DEFRA relevant to the assessment of these health outcomes determined the following:

- Incidence of vascular dementia (Andersson et al., 2018; Carey et al., 2018) – low quality of evidence for no effect of road traffic noise (noting that these studies found no association with risk of dementia, particularly after adjustment for air pollution). The more recent study (Cantuaria et al., 2021) that included 1,938,994 participants aged 60 years and older in Denmark found transportation noise (road and rail) to be associated with the incidence of all cause dementia and dementia subtypes, especially Alzheimer's disease.
- Dementia-related emergency admissions (Culqui et al., 2017; Linares et al., 2017) – very low-quality evidence for an effect from road traffic noise (noting that these studies indicate exposure to traffic noise may exacerbate symptoms of dementia).
- Cognitive assessment of dementia symptoms (Tzivian et al., 2016) – very low quality of evidence for an effect of road traffic noise (noting that this outcome is not changed with the inclusion of the additional studies from Tzivian et al. (2020) and Weuve et al. (2020)).
- Multiple sclerosis admissions (Carmona et al., 2018) – very low-quality evidence for an effect of road traffic noise (study indicates the potential for multiple sclerosis exacerbation from traffic noise).
- Parkinson's disease emergency admissions and healthcare (Díaz et al., 2018) – very low-quality evidence for an effect of road traffic noise (noting the study indicated that exposure to road traffic may exacerbate Parkinson's disease).

The DEFRA review concluded that the evidence for an effect of environmental noise on dementia and neurodegenerative outcomes is mixed. An association between dementia and road traffic noise was identified, however no associations have been identified for any other noise source. This conclusion remains largely unchanged with this review, noting that the Danish study (Cantuaria et al., 2021) provides exposure-response relationships for the risk of Alzheimer's disease for road and rail noise. While the Danish study is large, additional studies that provide consistent outcomes are required to provide robust evidence sufficient for the quantification of health impacts in other countries/settings. It is not appropriate at this time to include dementia (or other neurodegenerative diseases) in any quantitative assessment of health impacts from noise.

3.8.3 Birth weight and birth outcomes

The WHO evaluation of adverse birth outcomes (Nieuwenhuijsen et al., 2017) identified and evaluated (using the GRADE system) 14 studies published between June 2014 and December 2016 (including six studies on aircraft noise and five studies on road traffic noise). The WHO focused on more recent studies due to the quality of the older studies. No meta-analysis was undertaken due to the small number of studies identified in the WHO review. The WHO review found evidence of very low quality for associations between aircraft noise and pre-term birth, low birth weight, and congenital anomalies, and low-quality evidence for an association between road traffic noise and low birth weight, pre-term birth, and small for gestational age. Due to the low quality of evidence, no quantitative exposure-response relationships were recommended for the assessment of environmental noise, from any source.

The DEFRA review (ARUP, 2020; Clark et al., 2020) identified and evaluated (using the GRADE system) an additional seven studies published between January 2017 and March 2019. One of these studies related to noise generated from wind turbines (Poulsen et al., 2018) and is not specifically relevant to this review. Most of the other studies related to road traffic noise. The studies identified included those identified in the literature review conducted for this assessment.

This review also identified one additional study related to the assessment of birth outcomes and aircraft noise (Argys et al., 2020). One of the studies included in the evaluation of obesity (section 3.5) (Weyde et al., 2018) evaluated BMI in children where road noise exposures occurred during pregnancy or childhood. Both these studies have been included in this review.

The studies reviewed by DEFRA (using the GRADE system) were considered to have a low risk of bias. The additional study has also been reviewed using the GRADE system (refer to Appendix C**)** and considered to have a low to moderate risk of bias.

The studies identified considered a range of birth outcomes including pre-term birth, low birth weight, small for gestational age, as well as BMI later in life. Other effects evaluated included febrile seizures and congenital abnormalities. One study evaluated effects on male fertility. In relation to these outcomes, the DEFRA review determined the following (noting inclusion of the two additional studies identified in this evaluation).

Table 3.5 Summary of the strength of evidence for birth and reproductive outcomes for transport noise sources, post WHO (2018) evaluation (modified from (ARUP, 2020; Clark et al., 2020))

Health outcome	Quality of evidence and assessment of effect			References
	Aircraft noise	Road traffic noise	Railway noise	
Low birth weight	Very low quality – some effect (1 study)*	High quality – no effect (3 studies) (plus 1 additional study identified with no change to DEFRA evaluation)	Very low quality – no effect (1 study)	(Argys et al., 2020; Dzhambov et al., 2019; Smith et al., 2017; Wallas et al., 2019; Weyde et al., 2018)
Pre-term birth	N/A	Moderate quality – no effect (1 study)	N/A	(Wallas et al., 2019)
Small for gestational age	N/A	Moderate quality – no effect (2 studies)	Very low quality – no effect (1 study)	(Dzhambov et al., 2019; Smith et al., 2017)
Congenital abnormalities	N/A	Low quality – no effect (1 study)	N/A	(Pedersen et al., 2017)
Febrile seizures	N/A	Low quality – effect identified (1 study)	N/A	(Hjortebjerg et al., 2018)
Male fertility	N/A	Low quality – effect identified # (1 study)	N/A	(Min & Min, 2017)

* Based on evaluation of additional study identified in this review, and consideration of how this changes the quality of evidence presented by Nieuwenhuijsen et al. (2018), refer to Appendix A.

It is noted that a letter to the editor in relation to this paper (Dzhambov, 2017) requested further clarification in relation to the characterisation of exposure, noting that the outcomes appear to be consistent with other studies indicating that testosterone suppression could be underlying the effects seen.

Further to the above evaluation, Dzhambov and Lercher (2019a) conducted a systematic review and meta-analysis of the evidence in relation to road traffic noise and birth outcomes, specifically low birth weight, small for gestational age, and pre-term birth, based on studies published to May 2019. This analysis is considered to be an update of the WHO evaluation, noting that the WHO evaluation did not include meta-analysis. This review determined a moderate quality of evidence for an effect of maternal exposure from road traffic noise on low birth weight (and an association was identified and quantified as −8.26 g [−20.61, 4.10] lower birth weight with a 10 dB increase in L_{den}), and a very low quality of evidence of other effects, where no association was identified.

The analysis by Dzhambov and Lercher (2019) identified an effect for low birth weight, however the reviews by both the WHO (Nieuwenhuijsen et al., 2017) and DEFRA (ARUP, 2020) determined no effect. There are some differences in the approaches used in these reviews to determine the quality and weight of evidence. The DEFRA review considered the analysis from Dzhambov and Lercher (2019) and concluded that while the evidence may be considered equivocal, there is no harmful effect of road traffic noise on birthweight. There is no evidence in the current literature that changes this outcome.

On this basis, the available evidence does not support the assessment of birth outcomes from exposure to transport noise.

3.8.4 Cancer

Cancer was not a specific health outcome evaluated in the studies underpinning the WHO (2018) review, however more recent studies are available that have included review of cancer health outcomes in terms of noise exposures. The DEFRA review included an assessment of studies relating to cancer published in the period January 2014 to March 2019 (ARUP, 2020; Clark et al., 2020). The review identified 11 studies, three of which were excluded (as they did not measure or assess noise exposures). Of the eight studies evaluated, seven were conducted in Denmark, with six of the eight studies from a large Danish Diet, Health and Cancer longitudinal study. DEFRA has evaluated each of these in detail, using the GRADE system. The studies reviewed included studies identified in this review. These studies address a range of different cancers that include breast cancer, colorectal cancer, prostate cancer, and non-Hodgkin lymphoma. The risk of bias has been determined to be low for the eight studies included in the DEFRA analysis.

Further to the above, one additional study has been identified (Sørensen et al., 2021) in this review that relates to road and railway noise sources and breast cancer. This was a population cohort study in Denmark. The study has been reviewed (refer to Appendix C) and the risk of bias determined to be low.

Review of the quality of evidence by DEFRA relevant to the assessment of these health outcomes determined the following.

3.8.4.1 Breast cancer

One study was available for the assessment of aircraft noise (Hegewald et al., 2017), where the quality of evidence for an effect on the incidence of breast cancer was determined to be low.

Three studies were available for the assessment of road traffic noise (Andersen et al., 2018; Hegewald et al., 2017; Sørensen et al., 2014), however these had inconsistent findings and the quality of evidence of an effect was determined to be low.

Two studies were available for the assessment of railway noise (Hegewald et al., 2017; Sørensen et al., 2014), however these had inconsistent findings and the quality of evidence of an effect was determined to be low.

The additional study identified in this review (Sørensen et al., 2021) related to road and railway noise sources and the incidence of breast cancer in the Danish population. This study identified associations between exposure to road traffic noise (and rail noise, but to a less significant extent) and breast cancer incidence, with the stronger associations relating to exposures at the least noise impacted façade (ie, the quieter part of the building). All the available studies relating to breast cancer, however, provide inconsistent results. Hence, while this study adds to the available information, insufficient consistent data is available to change the overall quality of evidence from being low.

3.8.4.2 Colorectal cancer

One study is available that evaluates exposure to road traffic and rail noise sources (Roswall et al., 2017) where the overall quality of evidence (for an effect from road noise and no effect from rail noise) was determined to be low. No studies are available on aircraft noise.

3.8.4.3 Prostate cancer

One study is available that evaluates exposure to road traffic and rail noise sources (Roswall et al., 2015) where the overall quality of evidence (for no effect from road noise and no effect from rail noise) was determined to be low. No studies are available on aircraft noise.

3.8.4.4 Non-Hodgkin lymphoma

One study is available that evaluates exposure to road traffic noise sources (Sørensen et al., 2015) where the overall quality of evidence for an effect was determined to be low. No studies are available on aircraft or rail noise sources.

3.8.4.5 Cancer mortality

Two studies evaluated the incidence of cancer mortality (Roswall et al., 2016; Roswall et al., 2017) with high quality of evidence identified for no effects from road traffic noise on cancer mortality.

Overall, the data suggests some evidence of an effect from transport noise sources on some cancer outcomes. The data is heavily biased to the Danish population and relates to the incidence of some cancers. For some outcomes the evidence is inconsistent, and there are limited numbers of studies for many of the cancer outcomes evaluated. The limited data available on cancer mortality has not identified an effect or association with road traffic noise. On this basis, it is not recommended that cancer outcomes be quantified for population health impact assessments from transport noise.

3.8.5 Mental health and quality of life

The WHO has undertaken a review of environmental noise and a range of outcomes relating to mental health, quality of life, and wellbeing (Clark et al., 2018a) based on studies published from January 2005 to October 2015. The review identified 29 predominantly cross-sectional studies and used the GRADE system of review. Overall, most evidence was rated as very low quality, with evidence of effects only being observed for some noise sources and outcomes. Specifically, there are few studies of clinically significant mental health outcomes (harmful effects of emotional and conduct disorders in children), however more studies are needed.

Table 3.6 to Table 3.8 provide summaries of the outcomes of the WHO review in relation to the various areas of evaluation, with comparison against outcomes derived from studies published after the WHO review. The lack of evidence for noise effects across studies for many of the quality of life, wellbeing, and mental health domains examined does not necessarily mean that there are no effects: rather, that they have not yet been studied robustly for different noise sources.

DEFRA conducted a detailed review of studies relating to mental health and quality of life published in the period October 2015 to March 2019 (ARUP, 2020; Clark et al., 2020). The review identified 24 studies conducted in Europe (Belgium, Bulgaria, Germany, The Netherlands), United Kingdom, Scandinavia (Finland, Norway, and Sweden), Canada, South Korea, and New Zealand. The review included an evaluation of the quality of evidence using the GRADE system. The studies identified included a number of studies identified in this review. For many of the studies, the risk of bias was indicated to be unclear (due to a lack of information relating to the key areas evaluated: exposure assessment, confounding, selection of participants, health outcome assessment, not blinded outcome assessment[4]). The other studies were considered to have a low risk of bias, except one which was considered to have a high risk of bias (ARUP, 2020; Clark et al., 2020).

The studies identified and reviewed by DEFRA related to a range of health outcomes including self-reported quality of life or health, self-reported depression, anxiety and psychological symptoms, interview measures of depressive and anxiety disorders, emotional and conductive disorders in children, hyperactivity, wellbeing, and Attention Deficit Hyperactivity Disorder (ADHD).

Table 3.6 to Table 3.8 provide summaries of the outcomes of the review conducted by DEFRA. The DEFRA evaluation did not provide any updated meta-analysis of any of the health outcomes evaluated. In relation to the study conducted in New Zealand (Welch et al., 2018), this was a small study that found that noise-sensitive individuals had significantly poorer health-related quality of life when living near an airport.

This study identified an additional 15 studies published between 2015 and March 2021. Eight of these studies have been excluded from this review as they did not measure noise exposures in relation to the relevant health outcomes evaluated. Most of these excluded studies evaluated the association between noise annoyance and mental health or other quality of life indicators.

The additional studies included in this review relate to the impact of short-term effects of traffic noise on suicide rates and emergency department admissions for anxiety and depression (Díaz et al., 2020), effect of aircraft noise on psychological health (Baudin et al., 2018a) and the effect of road traffic noise on depression (depressive symptoms and antidepression medication use) (Orban et al., 2016). Four studies were identified that provided systematic reviews and meta-analysis:

- Schubert et al. (2019) provided a systematic review and meta-analysis of studies relating to transport noise exposure and the mental wellbeing (as behavioural and emotional disorders) of children and adolescents, based on studies published to February 2019.
- Dzhambov and Lercher (2019b) provided an update of the WHO evaluation of effects of road traffic noise on depression and anxiety, based on studies published to August 2019.
- Hegewald et al. (2020) provided a systematic review and meta-analysis of traffic noise and mental health including risk of depression and anxiety based on studies published to December 2019.
- Lan et al. (2020) provided a systematic review of evidence and meta-analysis of data relating to transportation noise and anxiety, based on studies published to February 2020.

3.8.5.1 Meta-analysis

The meta-analysis by Schubert et al. (2019) included 10 studies that evaluated the effects of traffic noise on the mental health of children. Seven of the studies evaluated noise exposure at school where the results found that aircraft noise at school was associated with hyperactivity/inattention, and road noise at school was

[4] Human behaviour is influenced by what we know or believe. In research there is a particular risk of expectation influencing findings, most obviously when there is some subjectivity in assessment, leading to biased results. Blinding (sometimes called masking) is used to try to eliminate such bias. In epidemiological studies blinding commonly refers to keeping the identification of cases and controls secret or preventing/minimising knowledge of the exposure being evaluated in surveys to prevent bias in outcomes.

associated with conduct problems. Three of the studies related to exposure to noise at home. Meta-analysis was undertaken on data from three studies for assessing the risk of childhood behavioural problems (as abnormal behavioural scores in standardised tests). This limited analysis (based on only a few exposure-response effect estimates) shows a statistically significant association between road traffic noise, hyperactivity/inattention, and total difficulties in children. The small number of suitable studies limits the strength of evidence (with the authors noting there are too few studies to evaluate publication bias).

The meta-analysis by Dzhambov and Lercher (2019) included 10 studies that evaluated the effect of road traffic noise on depression and anxiety in adults. This included a number of studies published up to August 2019 and identified in the DEFRA review, as well as a number of publications relating to a large study on neighbourhood characteristics and depression in the Netherlands (Generaal et al., 2019). The meta-analysis found an increased risk for depression (4%) and anxiety (12%), however the associations were not found to be statistically significant.

The meta-analysis by Hegewald et al. (2020) evaluated studies from all sources of transportation noise and mental health outcomes as psychological complaints and disorders (mild cognitive disorder, depressive episodes, anxiety disorders) including self-reported data, prescribed medications, and validated screening tools. The study included effects in children and adults, published to December 2019. The review identified 20 studies relating to depression and 11 studies relating to anxiety which include studies reviewed by the WHO and DEFRA. However, it is noted that some of the studies differed from those included in the Dzhambov and Lercher (2019b) review. In relation to depression, an association was identified for road traffic and rail noise sources, but the pooled estimates (exposure-response functions) were not found to be statistically significant. However, the association for aircraft noise was found to be statistically significant (from five studies). In relation to anxiety disorders, six studies were identified and used in the meta-analysis. Three studies related to road traffic sources, with one study also addressing aircraft noise and two studies also addressing rail noise. For road traffic noise, an association was identified although the pooled estimate was not found to be statistically significant. Insufficient data was available for aircraft and rail noise.

The meta-analysis by Lan et al. (2020) evaluated studies published to February 2020, and included those relating to road, rail, and/or air traffic sources and effects on anxiety measured using validated scales, questionnaires (self-reported), or medication use. This systematic review included those identified in earlier reviews by DEFRA and WHO, the meta-analyses discussed above (in relation to anxiety), and the additional studies identified in this review. The review used the GRADE system for evaluating the studies and the quality of evidence. The study by Díaz et al. (2020) was excluded as sampling was not undertaken on an individual level, and for this reason this study has not been further evaluated in this review. For the nine individual studies included in the meta-analysis, five were found to have a low risk of bias, with the other studies found to have a moderate risk of bias. No studies were identified with a high risk of bias. The overall quality of evidence was determined to be very low in relation to road, aircraft, and mixed traffic noise sources and anxiety, with the quality of evidence for rail noise determined to be low. Where all traffic noise sources are combined, the meta-analysis found a near significant association with 9% higher odds of anxiety associated with a 10 dB increase in L_{den}. A stronger association was reported for anxiety when based on medication use and diagnosis data. Traffic noise was more likely to be significantly associated with more severe anxiety. Meta-analyses for specific transport types revealed discrepancies in the effects of different noise sources, but none of them were significant.

In relation to the mechanisms by which exposure to noise adversely affects mental health, this is explained by stress and behavioural processes. The stress-diathesis hypothesis suggests that transportation noise, as an environmental stressor, can increase physiological arousal and stress hormone secretion (eg, adrenaline and cortisol) through repeated stimulation of the endocrine system and autonomic nervous system. Prolonged activation of these responses may cause mental disorders including anxiety. According to the behavioural mechanism, it emphasises that people proactively deal with exposure to noise by adjusting their behaviour in noisy conditions to reduce exposure through the appraisal of noise (in terms of danger, loss of quality, the meaning of the noise, challenges for environmental control, etc) and coping strategies. As a result, actively coping with noise may be sufficient to mitigate the ill effects (Lan et al., 2020).

Outcomes of the meta-analysis studies detailed above are also included in summary tables, Table 3.6 to Table 3.8.

Table 3.6 Comparison of the strength of evidence for the assessment of mental health, wellbeing, and quality of life outcomes – aircraft noise (modified from (ARUP, 2020; Clark et al., 2020))

<table>
<tr><th rowspan="2">Health outcome</th><th colspan="3">Quality of evidence and assessment of effect</th></tr>
<tr><th>WHO review to 2015 (Clark et al., 2018b)</th><th>DEFRA review 2015 to 2019 (ARUP, 2020; Clark et al., 2020)</th><th>Further meta-analysis to 2019 or 2020</th></tr>
<tr><td>Self-reported quality of life on health</td><td>Very low quality – no effect</td><td>Very low quality – no effect</td><td>N/A</td></tr>
<tr><td>Medication intake for treatment of anxiety and depression</td><td>Very low quality – harmful effect</td><td>N/A</td><td rowspan="3">Low quality – statistically significant effect for depression, non-significant effect for anxiety</td></tr>
<tr><td>Self-reported depression, anxiety, and psychological symptoms</td><td>N/A</td><td>N/A</td></tr>
<tr><td>Interview measures of depressive and anxiety disorders</td><td>Very low quality – harmful effect</td><td>Low quality – harmful effect</td></tr>
<tr><td>Emotional and conduct disorders in children</td><td>Low quality – no effect</td><td>N/A</td><td>N/A</td></tr>
<tr><td>Hyperactivity</td><td>Low quality – harmful effect</td><td>N/A</td><td>N/A</td></tr>
<tr><td>Wellbeing</td><td>Not evaluated</td><td>Very low quality – harmful effect</td><td>N/A</td></tr>
</table>

Table 3.7 Comparison of the strength of evidence for the assessment of mental health, wellbeing, and quality of life outcomes – road traffic noise (modified from (ARUP, 2020; Clark et al., 2020))

<table>
<tr><th rowspan="2">Health outcome</th><th colspan="3">Quality of evidence and assessment of effect</th></tr>
<tr><th>WHO review to 2015 (Clark et al., 2018b)</th><th>DEFRA review 2015 to 2019 (ARUP, 2020; Clark et al., 2020)</th><th>Further meta-analysis to 2019 or 2020</th></tr>
<tr><td>Self-reported quality of life on health</td><td>Low quality – no effect</td><td>N/A</td><td>N/A</td></tr>
<tr><td>Medication intake for treatment of anxiety and depression</td><td>Very low quality – no effect</td><td>Very low quality – harmful effect</td><td rowspan="3">Low quality – non-significant effect for anxiety and depression</td></tr>
<tr><td>Self-reported depression, anxiety, and psychological symptoms</td><td>Very low quality – no effect</td><td>Very low quality – no effect</td></tr>
<tr><td>Interview measures of depressive and anxiety disorders</td><td>Very low quality – no effect</td><td>Low quality – harmful effect</td></tr>
<tr><td>Emotional and conduct disorders in children</td><td>Moderate quality – harmful effect</td><td>Low quality – harmful effect</td><td>N/A</td></tr>
<tr><td>Hyperactivity</td><td>Moderate quality – harmful effect</td><td>Low quality – harmful effect</td><td>N/A</td></tr>
<tr><td>Cortisol on children</td><td>N/A</td><td>Very low quality – harmful effect</td><td>N/A</td></tr>
<tr><td>Wellbeing</td><td>Not evaluated</td><td>N/A</td><td>N/A</td></tr>
<tr><td>ADHD in children</td><td>Not evaluated</td><td>Low quality – harmful effect</td><td>N/A</td></tr>
</table>

Table 3.8 Comparison of the strength of evidence for the assessment of mental health, wellbeing, and quality of life outcomes – railway traffic noise (modified from (ARUP, 2020; Clark et al., 2020))

Health outcome	Quality of evidence and assessment of effect		
	WHO review to 2015 (Clark et al., 2018b)	DEFRA review 2015 to 2019 (ARUP, 2020; Clark et al., 2020)	Further meta-analysis to 2019 or 2020
Self-reported quality of life on health	Low quality – harmful effect	N/A	N/A
Medication intake for treatment of anxiety and depression	N/A	Very low quality – harmful effect	Low quality – non-significant effect for anxiety, insufficient data for depression
Self-reported depression, anxiety, and psychological symptoms	N/A	Very low quality – no effect	
Interview measures of depressive and anxiety disorders	N/A	Low quality – harmful effect	
Emotional and conduct disorders in children	Moderate quality – harmful effect	N/A	N/A
Hyperactivity	Moderate quality – no effect	N/A	N/A
Wellbeing	N/A	N/A	N/A

Overall, additional studies are available since publication of the WHO (2018) review in relation to mental health, wellbeing, and quality of life outcomes. Most of the available studies relate to mental health measures, however there is considerable variability in the available studies, with many adopting different measures for assessing depression and anxiety. While an association between transport noise and mental health outcomes has been identified in a number of studies, few statistically significant exposure-response relationships have been identified that would be sufficiently robust to include in any quantitative analysis.

If mental health were considered a key health outcome for an assessment, the most robust outcome relates to the relationship for the impact of all transport noise sources combined (per 10 dB increased in noise) on anxiety from Lan et al. (2020). This would need to be assessed based on data on medication use and diagnosis of anxiety disorders. These are not data that are routinely reported or available in New Zealand and hence while the studies may be able to quantify such outcomes, it is impractical to apply these in the New Zealand population due to a lack of data.

In relation to wellbeing and quality of life indicators, insufficient new data is available to indicate a change in the WHO (2018) evaluation and no quantitative evaluation of these health outcomes is recommended.

3.9 Recommendations

On the basis of the health review completed, the following exposure-response functions are recommended for use in the quantification of health impacts of transport noise on the New Zealand population. While it is noted that many of the recommended exposure-response relationships are consistent with those recommended by the WHO (2018), some have been modified based on more recent evaluations. For road traffic noise and its effects on the cardiovascular system (as incidence of IHD), different exposure-response relationships are recommended for the assessment of urban and rural areas to address potential differences in noise sensitivity in more quiet rural areas (consistent with outcomes of New Zealand-specific studies). Insufficient data is available to provide urban and rural relationships for other transport sources or health outcomes.

Table 3.9 Summary of recommended exposure-response relationships for the assessment of health impacts of transport noise in New Zealand

Health outcome	Noise metric	Lowest level of exposure* (dB)	Exposure-response relationship per 10 dB increase (RR = relative risk or OR = odds ratio) [95% confidence interval]	Quality of evidence
Road traffic noise				
Cardiovascular effects				
Incidence of ischaemic heart disease (IHD) – urban areas	L_{den}	53	RR = 1.02 [1.0–1.04]	High
Incidence of ischaemic heart disease (IHD) – rural areas	L_{den}	53	RR = 1.08 [1.01–1.15]	High
Diabetes (incidence of Type 2 diabetes)	L_{den}	N/A	RR = 1.11 [1.08–1.15]	Low
Annoyance (as % highly annoyed, %HA)	L_{den}	40	OR = 3.03 [2.59–3.55] %HA = 78.9270–3.1162 × L_{den} + 0.0342 × L_{den}^2	Moderate
Sleep disturbance (as % highly sleep disturbed, %HSD)	L_{night}	43	OR = 2.13 [1.82–2.48] %HSD = 19.7885–0.9336 × L_{night} + 0.0126 × L_{night}^2	Moderate
Railway noise				
Cardiovascular effects – incidence of IHD	L_{den}	N/A	RR = 1.01 [0.99–1.03]	High
Annoyance (%HA)	L_{den}	34	OR = 3.53 [2.83–4.39] %HA = 38.1596–2.05538 × L_{den} + 0.0285 × L_{den}^2	Moderate
Sleep disturbance (%HSD)	L_{night}	33	OR = 3.06 [2.38–3.93] %HSD = 67.5406–3.1852 × L_{night} + 0.0391 × L_{night}^2	Moderate
Aircraft noise				
Cardiovascular effects – incidence of IHD	L_{den}	47	RR = 1.09 [1.04–1.15]	Very low
Annoyance (as % highly annoyed, %HA)	L_{den}	33	OR = 4.78 [2.28–10.05] %HA = −50.9693 + 1.0168 × L_{den} + 0.0072 × L_{den}^2	Moderate
Aircraft noise – sensitivity analysis	L_{den}	33	%HA = −9.199*10^{-5} (L_{den}-42)^3 + 3.932*10^{-2} (L_{den}-42)^2+ 0.2939 (L_{den}-42)	Moderate
Cognitive impairment (as reading and oral comprehension)	L_{den}	55	1 to 2-month delay per 5 dB increase	Moderate
Sleep disturbance (as % highly sleep disturbed, %HSD)	L_{night}	35	OR = 1.94 [1.61–2.33] %HSD = 16.7885–0.9293 × L_{night} + 0.0198 × L_{night}^2	Moderate

*Note that these are external free-field equivalent noise levels incident on the façade of the dwelling.

4 GIS processing and noise modelling

4.1 Overview

A 3D noise model of the existing environment surrounding the study road and rail alignments was prepared and calculated in SoundPLAN (v8.2). The current population's noise exposure was derived from the results of the noise modelling, so that the health and cost impacts across New Zealand could be identified and visualised.

This section focuses on the development of the transport noise exposure model and preparation of the model outputs for visualisation and interpretation. This has been carried out in general accordance with the methodology set out in section 2.1.

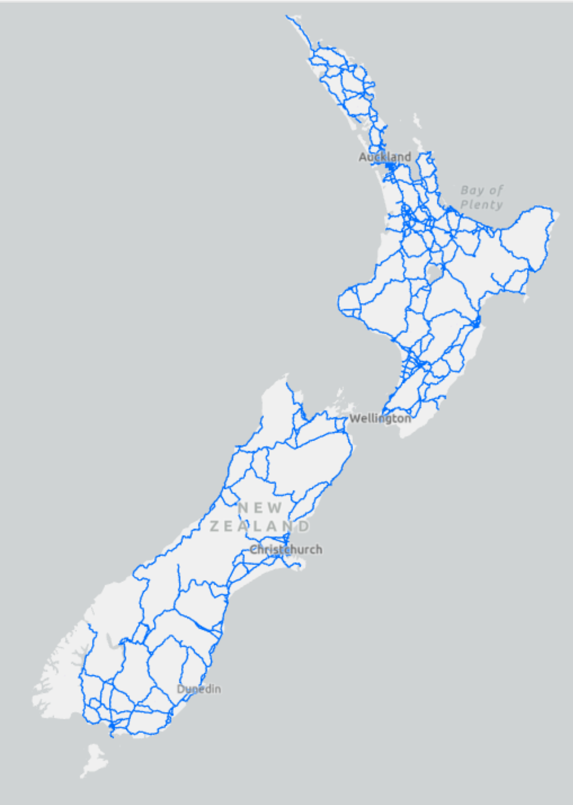
Note that separate levels for road and rail noise were calculated at receivers since cumulative effects were not considered, therefore receivers affected by both road and rail noise were included in both models.

4.2 Preparation of model inputs

4.2.1 Study area

Figure 4.1 shows the extent of state highways and the arterial road and rail network across the North and South Island that was modelled.

Figure 4.1 Road and rail network (left and right respectively) over the New Zealand North and South Island



An assessment area of 600 m either side of the road and rail corridors was used during the modelling exercise. This study area was chosen as it would allow all health indicators to be captured in the results, whereas with a smaller assessment area (eg, 100 m), the noise levels in the assessment area would not have fallen to the lowest band where health impacts would be expected.

The calculation of transport noise exposure was limited to dwellings within the nominated assessment distances. Dwellings were identified as detailed in the following section.

4.2.2 Modelling of the residential population

The number of people affected per dwelling by the road and rail network is a key parameter for calculating the health costs of noise. This required dwellings to be identified amongst garages, sheds, and other structures.

A methodology was developed to identify dwellings, after which census data was used to assign the number of people to each dwelling. The methodology is described in this section.

The following datasets were used in this methodology:

- Building Outline data (Land Information New Zealand (LINZ)), provided as polygons
- Primary Land Parcel data (LINZ), provided as polygons
- Street Address data, provided as address points (LINZ)
- 2018 Census Dwelling total New Zealand by Statistical Area 1
- 2018 Census Individual part 1 total New Zealand by Statistical Area 1
- GIS Open Street Map (OSM) land use a free 1 (OSM).

4.2.2.1 Methodology to determine dwellings

Various methods were tested for identifying dwellings within a given parcel. The final calculation methodology was chosen based on ease of understanding and calculation time.

The chosen methodology identified dwellings based on building outlines, parcels, and address points. Various factors needed to be accounted for, including:

- multiple dwellings within the same parcel (eg, subdivided sections)
- building outlines spanning over multiple parcels (eg, adjoining houses sharing a single roof)
- multiple dwellings within the same building outline (eg, apartment block).

Due to the factors listed above, the building outlines had to be manipulated to identify all dwellings within the study areas before the calculations could be run.

The calculations assumed all dwellings fell fully within each parcel. Because of this, the building outlines had to be split where they crossed over parcel boundaries. After this, a negative 1 m buffer was applied to the building outlines to ensure they were fully contained within each parcel.

The number of buildings and address points[5] that were located within each parcel was then calculated. This information was assigned to each building within each parcel, along with the unique identification (ID) number of the parcel.

The land use of the buildings was determined using statistical areas from OSM land use data. The OSM data was considered the most complete national scale dataset to use to define land use.

[5] Note that while address points were generally located within their corresponding parcel, most address points were not located within their corresponding building outline.

Benefits to using OSM included:

- OSM had land use data that classifies land into residential, commercial, industrial, retail, schools etc.
- OSM is a current and available dataset (maintained by the community) and can be manipulated with minimal labour.
- The use of OSM data is transparent (unrestricted access).
- OSM provides nationwide data using a single classification system.

The transfer of the land use type, total resident count, and total dwelling count from the OSM data to the building outlines was achieved using a function called a spatial join. This function copies attributes from one dataset to another based on the spatial relationship, for example, one dataset intersecting with the other.

It is noted that the OSM land use data is not complete, so there were statistical areas that had no land use classification particularly within small towns and regional areas. Buildings that were not allocated a land use were recorded as ‘Not Assigned’.

To identify residential buildings that did not have any OSM land use classification, that is, buildings that existed outside of the OSM data extent, it was necessary to determine which buildings were the largest and closest to the road. This is discussed further in the next section.

To determine the distances from buildings to the road and rail networks the ‘Near’ function in ArcGIS Pro was used. This function calculates distance for each object in one dataset from an object in another dataset. By using this function, a new field was added to the building outline that recorded the distance to road or rail from that building.

To identify the closest building to the road/rail network within each parcel, the data was filtered by calculated distance in ascending order. The building ID corresponding to the first occurrence of each parcel ID was then recorded in a summary table. In effect, this table contained the closest building to the road/rail network per parcel.

The census data contained statistical areas that had the total residential population and the total number of dwellings for each area. From this, a ratio was calculated to estimate the number of people per building per statistical area (total resident population/total number of dwellings). This ratio was assigned to all buildings within each statistical area.

The region and health district each building fell under was obtained via spatial join of those datasets obtained from LINZ. While this was not relevant to the methodology to determine dwellings, this information would later be used to quantify the cost results per relevant area.

After the above steps, a dataset was obtained of building outlines including the following information (attributes):

- Building ID
- Building Area
- Parcel ID
- Count of total address points in the parcel containing the building
- Count of total buildings in the parcel containing the building
- Whether the building was the largest building in the parcel (Yes/No)
- Whether the building was the nearest to the road/rail network (Yes/No)
- OSM Land Use classification (where available)
- Statistical Area ID for the statistical area in which the building was located
- Total dwelling count for the statistical area in which the building was located
- Total resident count for the statistical area in which the building was located

- Average number of people per dwelling per statistical area (Resident Population Count/Dwelling Count ratio)
- Region
- District
- Health District.

4.2.2.2 Classification rules to assign dwellings and number of people per building

A set of rules called attribute queries were run to determine which buildings could be classed as dwellings, and then the number of people was assigned to the building.

The classification rules assumed that the number of address points in each parcel would be equal to the number of dwellings in that parcel. For example, a parcel with three address points indicates that there are three dwellings associated with the parcel.

The attribute queries created the following fields for each building outline:

- Processed – field to be used to determine if the building was classed by a rule (Yes/No)
- Query string – field to record the query defining the classification rule
- Number of people per building.

The following set of rules were determined by testing to provide the best results. The rules were applied in sequential order.

Rule 1

Identify buildings where the:

- land use is residential
- building count in the parcel is one
- address point count is one.

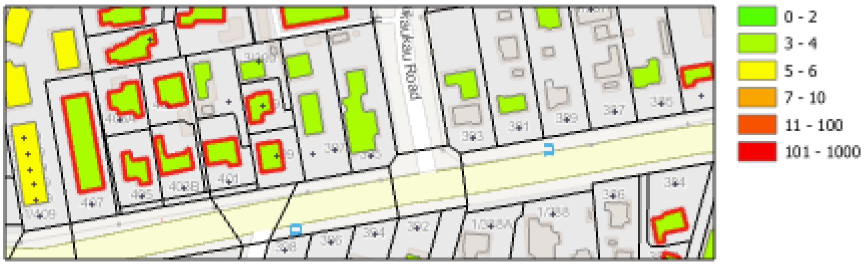
The selection returns buildings classed as residential where the parcel it is in contains only one building and one address point. It is assumed that this building would be the dwelling.

The people per building is then calculated as being the average number of people per dwelling per statistical area (from here referred to as ‘the ratio’ for conciseness). For example, taking 3.66 as the average number of people per dwelling:

Number of people = 3.66 * 1

The Processed field is updated to record that this selection of buildings has been processed. The Query string field is also updated to record the query that defined the rule.

Figure 4.2 Illustration of Rule 1, red outline identifies selected buildings



Rule 2

Identify buildings where the:

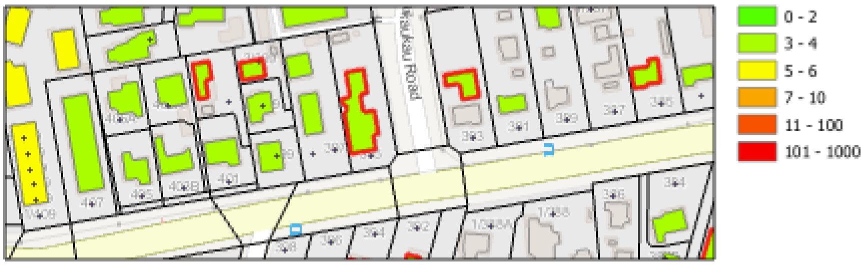
- land use is residential
- building count in the parcel is two
- address point count is one
- building is recorded as the largest building in the parcel.

The selection returns buildings which are classed as residential where the parcel contains two buildings. The largest building in the parcel is assumed to be the dwelling. An example of this is a parcel with a house and a shed.

The people per building is then calculated as being the ratio.

The Processed field is updated to record that this selection of buildings has been processed. The Query string field is also updated to record the query that defined the rule.

Figure 4.3 Illustration of Rule 2, red outline identifies selected buildings

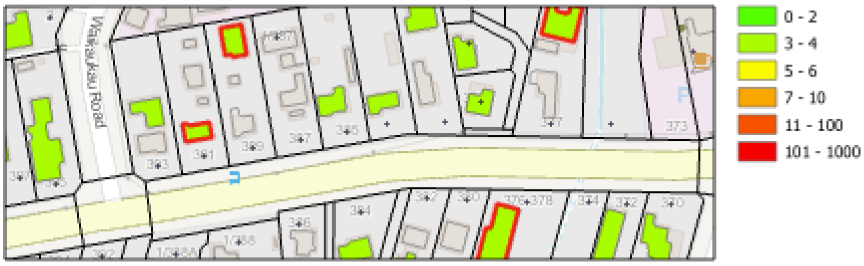


Rule 3

This rule applies the same logic outlined within Rule 2 with the exception that buildings are identified where the count within the land parcel is greater than two.

The selection returns buildings classed as residential where the parcel contains three or more buildings, and it is assumed that the largest building is the dwelling.

Figure 4.4 Illustration of Rule 3, red outline identifies selected buildings



Rule 4

Identify buildings where the:

- land use is residential
- building count in the parcel is one
- address point count is greater than one.

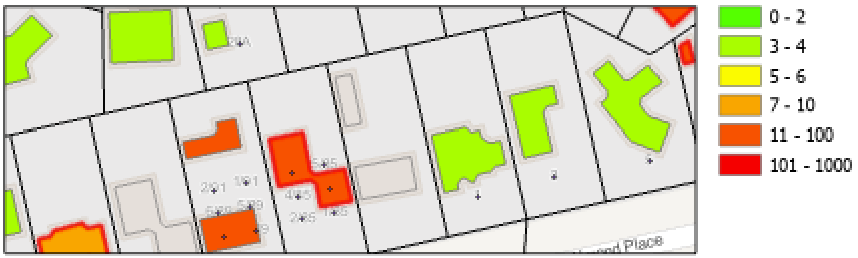
The selection returns buildings classed as residential where the parcel contains one building and the building has more than one address associated with it. It is assumed that this building would contain multiple dwellings. An example is an apartment block made up of one building containing multiple dwellings.

For this rule, the people per building is calculated as the address count multiplied by the ratio, for example:

Number of people = 3.66 * 50

The Processed field is updated to record that this selection of buildings has been processed. The Query string field is also updated to record the query that defined the rule.

Figure 4.5 Illustration of Rule 4, red outline identifies selected buildings



Rule 5

Identify buildings where the:

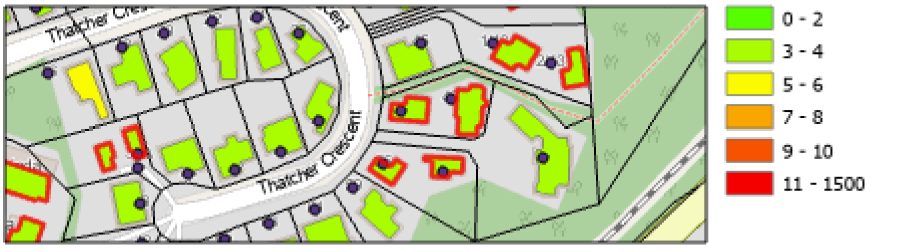
- land use is residential
- building count in the parcel is greater than zero
- building count in the parcel is equal to the address count in the parcel
- Processed field is equal to ‘No’ (ie, none of the previous rules have been applied).

The selection returns buildings classed as residential where the parcel contains the same number of buildings and address points, and the building has not been classified from any of the previous rules. It is assumed that each address on the parcel is a dwelling. An example is a subdivided parcel that contains more than one dwelling.

The people per dwelling is then set as the ratio.

The Processed field is updated to record that this selection of buildings have been processed. The Query string field is also updated to record the query that defined the rule.

Figure 4.6 Illustration of Rule 5, red outline identifies selected buildings



Rule 6

Identify buildings where the:

- land use is residential
- building count in the parcel is greater than or equal to the address count
- building area is greater than 50 m^2
- Processed field is equal to ‘No’ (ie, none of the previous rules have been applied).

The selection returns buildings classed as residential where there are an equal or greater number of buildings compared to address points in the parcel, and the building outline areas are greater than 50 m^2. An example is a residential complex that includes smaller outbuildings. In this case, buildings that are smaller than 50 m^2 are not classified as dwellings.

The people per building is then calculated as being the address point count in the parcel multiplied by the ratio, then divided by the number of buildings in the parcel.

The Processed field is updated to record that this selection of buildings has been processed. The Query string field is also updated to record the query that defined the rule.

Figure 4.7 Illustration of Rule 6, red outline identifies selected buildings

Rule 7

Identify buildings where the:

- land use is residential
- building count in the parcel is less than the address count
- building area is greater than 50 m^2
- Processed field is equal to ‘No’ (ie, none of the previous rules have been applied).

The selection returns buildings classed as residential where the parcel contains more address points than buildings. Buildings with an area less than 50 m^2 are excluded from the selection. An example is unit complexes where the complex is made up of multiple units. In this case the people per building is calculated as:

People in building = ratio * address count/building count

For example, number of people = 3.66 * 5 / 3

The Processed field is updated to record that this selection of buildings has been processed. The Query string field is also updated to record the query that defined the rule.

Figure 4.8 Illustration of Rule 7, red outline identifies selected buildings

Rule 8

Identify buildings where the:

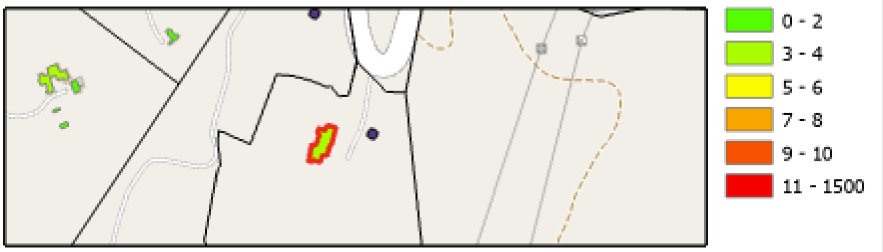
- land use is 'Not Assigned'
- building count in the parcel is one
- address count in the parcel is one
- building area is less than 1000 m^2
- Processed field is equal to 'No' (ie, none of the previous rules have been applied).

The selection returns buildings that are classed as 'Not Assigned' where the parcel contains one address and one building with an area less than 1000 m^2. The query identifies rural and remote properties that fell out of the OSM land use extent and therefore had no land use classification.

The people per building is then calculated as being the address count multiplied by the ratio.

The Processed field is updated to record that this selection of buildings has been processed. The Query string field is also updated to record the query that defined the rule.

Figure 4.9 Illustration of Rule 8, red outline identifies selected buildings



Rule 9

Identify buildings where the:

- land use is 'Not Assigned'
- building count in the parcel is greater than one
- address count in the parcel is one

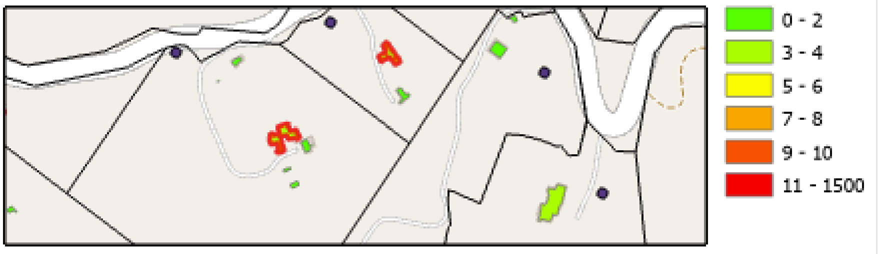
- building is the largest building in the parcel
- building area is less than 1000 m²
- Processed field is equal to 'No' (ie, none of the previous rules have been applied).

The selection returns buildings that are classed as 'Not Assigned' where the parcel contains one address, more than one building, outline area less than 1000 m², and that has not been classified from any of the previous rules. The query identifies rural and remote properties that fell out of the OSM land use extent that have more than one building, and identifies the building with the largest of outline as the dwelling.

The people per building is then calculated as being the address count multiplied by the ratio.

The Processed field is updated to record that this selection of buildings has been processed. The Query string field is also updated to record the query that defined the rule.

Figure 4.10 Illustration of Rule 9, red outline identifies selected buildings



Rule 10

Identify buildings where the:

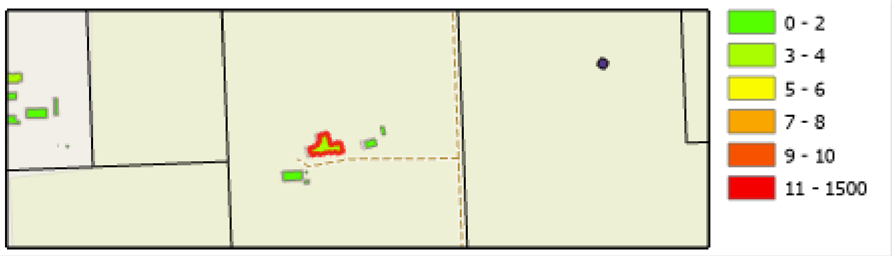
- land use is 'vineyard, farmland, farmyard, or forest'
- building count in the parcel is greater than one
- address count in the parcel is one
- building is the largest building in the parcel
- building area is less than 1000 m²
- Processed field is equal to 'No' (ie, none of the previous rules have been applied).

The selection returns buildings that are classed as 'vineyard, farmland, farmyard, or forest' where the parcel contains one address, more than one building, outline area less than 1000 m², and that has not been classified from any of the previous rules. The query identifies rural and remote properties that have more than one building and identifies the building with the largest of outline as the dwelling.

The people per building is the ratio.

The Processed field is updated to record that this selection of buildings has been processed. The Query string field is also updated to record the query that defined the rule.

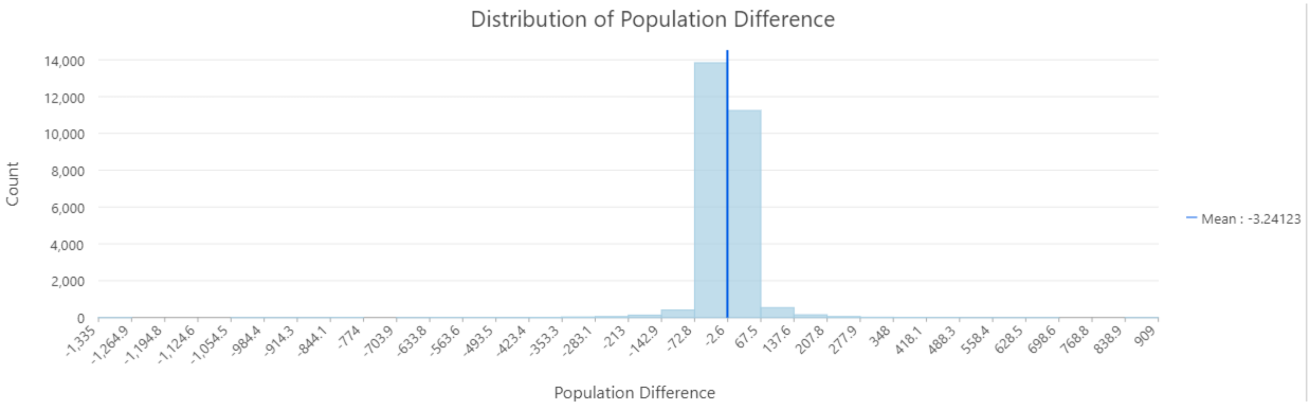
Figure 4.11 Illustration of Rule 10, red outline identifies selected buildings



4.2.2.3 Assessment of population calculation results

The accuracy of the allocation of people counts to dwellings was assessed by comparing the sum of all the people assigned to building outlines per statistical area against each original statistical area population total.

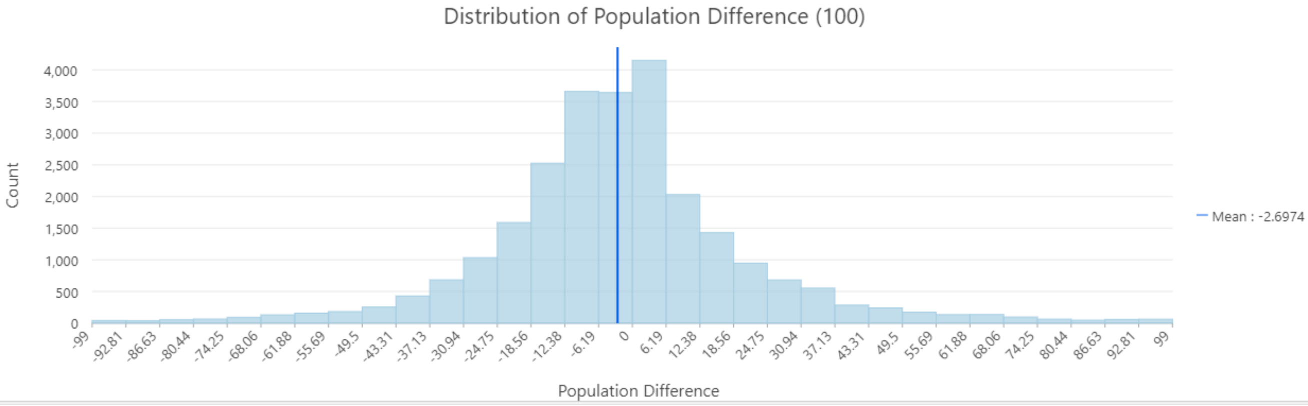
Figure 4.12 Comparison of the calculated sum of total population per Statistical Areas Level 1 (SA1) area versus the official current resident population for that area



The results indicate that the assignment of people counts per dwellings per statistical area has led to generally accurate overall population counts after assignments for the majority of cases. Due to the blanket approach which classified the outlines on a national scale, it was expected that there would be some error. The defined rules could not account for all the potential variations. Further work would be needed to manage the outliers and adjust the number of people assigned to those buildings manually.

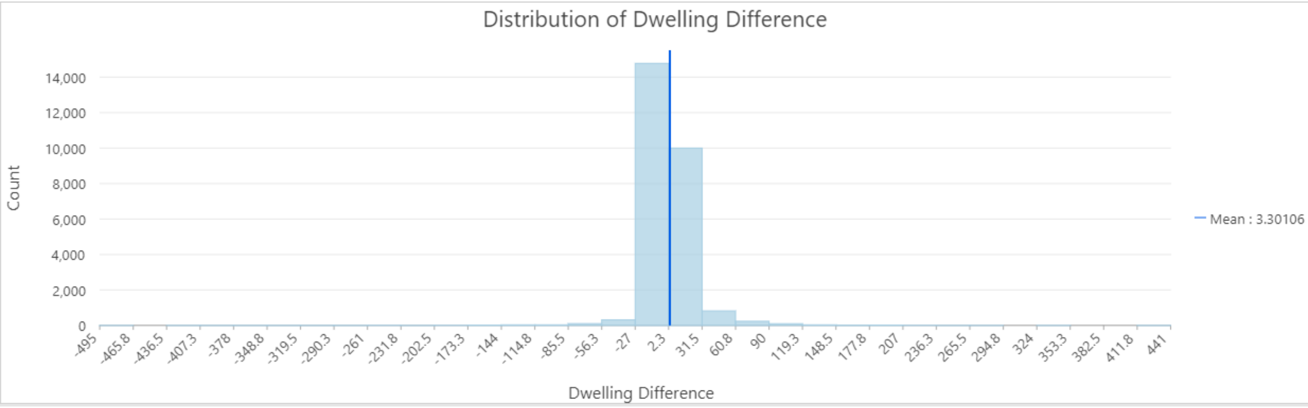
The graph below shows a restricted version of the comparison where population differences per area greater than 100 people have been removed. Out of the 26,313 Statistical Areas Level 1 (SA1) areas 25,648 (97%) showed a difference of less than 100 people.

Figure 4.13 Adjusted comparison of population difference excluding values greater than 100



A similar exercise was carried out for the total number of dwellings identified using the classification method. The difference between the number of buildings classified as dwellings per statistical area and the Stats NZ dwelling data for that area was compared. Figure 4.14 shows the result.

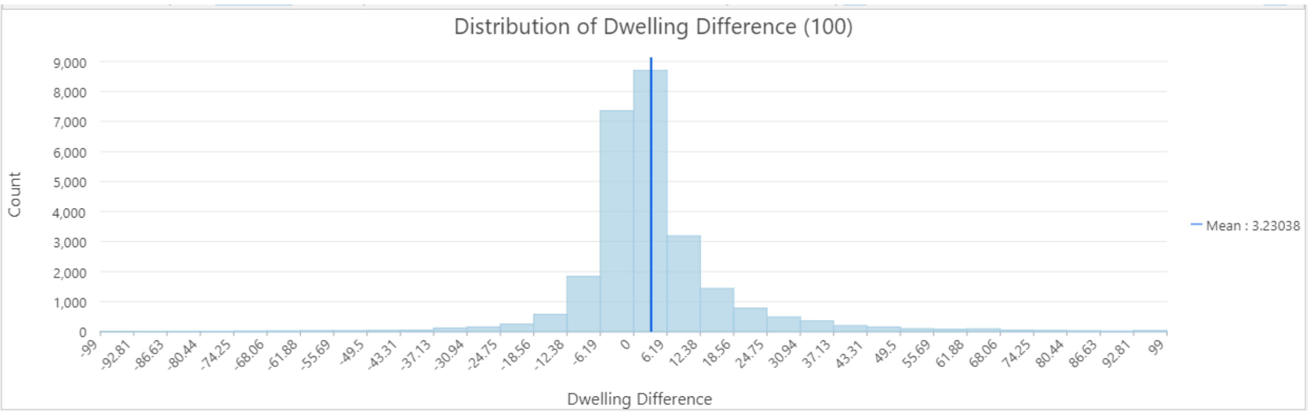
Figure 4.14 Comparison of population difference verses the Stats NZ data



As for the population counts, more work would be required to manage the outliers by visually inspecting areas that are returning large differences.

Excluding difference values greater than 100 shows the following result.

Figure 4.15 Adjusted comparison of the dwelling difference excluding values greater than 100

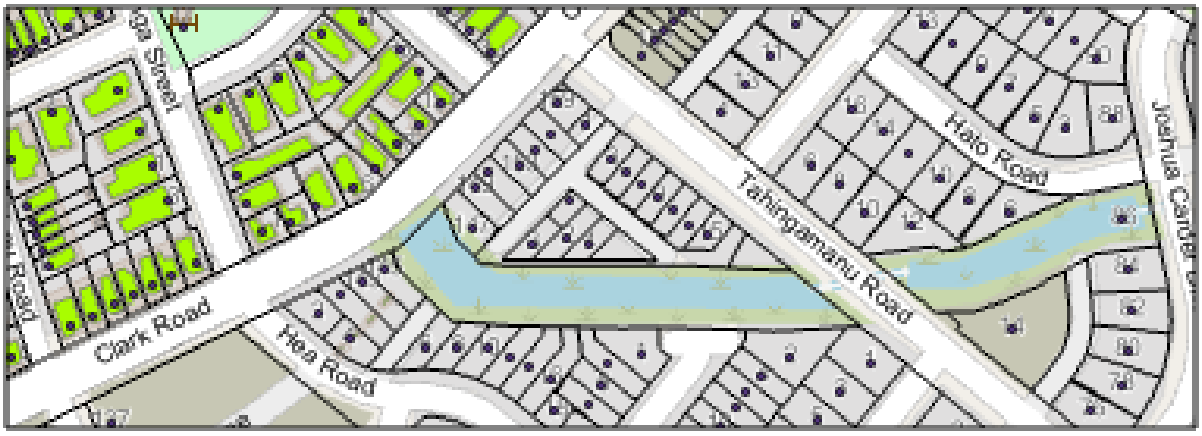


The methodology used has generally worked well for identifying dwellings within each parcel. Visually inspecting the results shows that in most cases the rules have correctly identified a residential building.

The number of people assigned to the building is based on the population dwelling ratio calculated from the total population and total number of dwellings for each statistical area, so is a derivative of the statistical data.

Some limitations are the coverage and recency of the building outline dataset. In some areas the data is out of date and does not include recent housing developments. An example is the Hobsonville suburb in Auckland, where significant development has occurred in the past 5 years. The aerial imagery from which the outlines were derived was flown in 2017 and does not reflect the new developments. This can be seen in Figure 4.16, where recorded building outlines are shown in green and empty grey parcels with address points should also contain building outlines.

Figure 4.16 Image showing areas of development that do not contain a building outline



Parcels with no building outlines were not considered further in this study, even if a dwelling now exists on the parcel, due to unavailability of reliable data. In further work it would be useful to apply a method or process to account for this scenario.

Figure 4.17 Series of images illustrating which buildings have been classified as a dwelling within a parcel

4.2.3 Data sources

The development of the transport noise exposure model required input data for:

- land geometry and features within 1200 m of the alignments
- road model attributes
- rail model attributes.

The full extent of attributes relevant to each dataset is summarised below:

Figure 4.18 Attribute descriptions

3D Rail Network

Id Number
Name of RailLine
TrackInfo
Trackline Type
Source Group
Direction of Travel
Emission_Calc
Maximum Posted Speed
Track TypeTrack
Cond_SP
BallastBridge Maximo ID
Bridge Present
Width Of Bridge
Bridge_thickness
Bridge TypeBridgeID_SP
Tunnel Maximo ID
Tunnel Present
Curve Start
Curve End
Radius of Curve
RadiusID_SP
Turnout Maximo IDTurnout_Crossing_SP
Station
Gradient

3D Road Network

Structure Type
Classification
Speed
Width
Heavy Vehicle percentage
AADT
Number of lanes
Surface Type
Chip Size
Rt (Adjustment value for trucks)
Rc (Adjustment value for trucks)

3D Building Footprints

Building ID
Floor Elevation
Roof Elevation
Building Height
Area

The above data was consolidated into the Esri shapefile format (.shp), standardised using the NZGD2000 based New Zealand Transverse Mercator projection to allow efficient data transfer into SoundPLAN. The preparation of road, rail, building, and terrain data in GIS software streamlined the input of data into SoundPLAN noise modelling software.

A summary of the data sources is provided in Table 4.1.

Table 4.1 Data sources

Data	Description	Source
General project geometry	Topography, buildings, and road alignments	Abley and LINZ
Road model attributes	Traffic volumes, speeds, and road surface types	NZTA and Abley
Rail model attributes	Track type, train volumes, and track features	KiwiRail

All the received project input data was stored in a central AECOM database in the appropriate file format, such as Excel, PDF, shapefiles or other, as provided. All the received spatial data was converted, where necessary, and stored in an Esri shapefile format, standardised using the NZGD2000 based New Zealand Transverse Mercator projection.

Details for each dataset are further discussed and identified in the following sections.

4.2.3.1 Building outlines

Digital building outlines within the 600 m assessment distance include the assigned attributes as set out in Table 4.2.

Table 4.2 Buildings data

Road data attribute	Description/Dataset particulars
All buildings within 600 m of road edge	2D Polygon shapefile, obtained from LINZ NZ Buildings Outline and other sources
Building elevation	Building elevation set to mean height of the terrain below the building
Unique ID	Unique ID for identification purposes
Parcel number	Included parcel boundaries when possible
Street name	Street address (text)
Street number	Address number (text)
Full address	Combination of the street number and street name
Type	Residential/temporary accommodation/education/healthcare/retirement/marae/commercial/industrial/mixed use
Building height	Set at 6 metres for residential and 8 metres for commercial and industrial buildings. (Refer to section 4.2.3.3)

The building dataset was based on the LINZ 'NZ building outlines', extracted from the LINZ data service (last updated 24 August 2020).

4.2.3.2 Topography

A consistent methodology for modelling of terrain was applied across all regions of New Zealand. The following options were considered:

1. Use the Esri world terrain data to extract a 1200 m wide corridor of 25 m grid resolution elevation data along the network. This would provide a consistent elevation base for the project, however the elevation data did not model the bare ground heights. This resulted in many areas that included above ground features such as trees and other large structures. This was considered a fatal flaw as these structures would act as barriers that would detrimentally affect the resulting sound contours. An effort was made to mask out these above ground elements by trying to identify trees and large areas of bush using remote sensing techniques, however the results were not accurate or consistent enough to warrant adoption.
2. Use a mixture of the most recently available LIDAR elevation data from councils across New Zealand and fill the remaining areas of no coverage with Topo 50K contours (20 m interval). This approach did take advantage of the available high resolution (1 m) elevation LIDAR data but had problems in large, flat no coverage areas where there was little or no 20 m contour information within the 1200 m corridor. Being unable to model the terrain for these areas and having potentially 19 m variations between contour lines made the approach unsuitable.
3. Retain the accuracy of the most recently available LIDAR data and supplement the missing areas with a national Digital Elevation Model (DEM)[6] that could provide an elevation value at a grid spacing that would ensure the 1200 m corridor had enough data to build a terrain even on flat ground. The two DEM options considered were the National 8 m Elevation DEM and the New Zealand School of Surveying (NZSoS) 15 m nationwide DEM. Although the 8 m DEM was a higher resolution, it is primarily derived

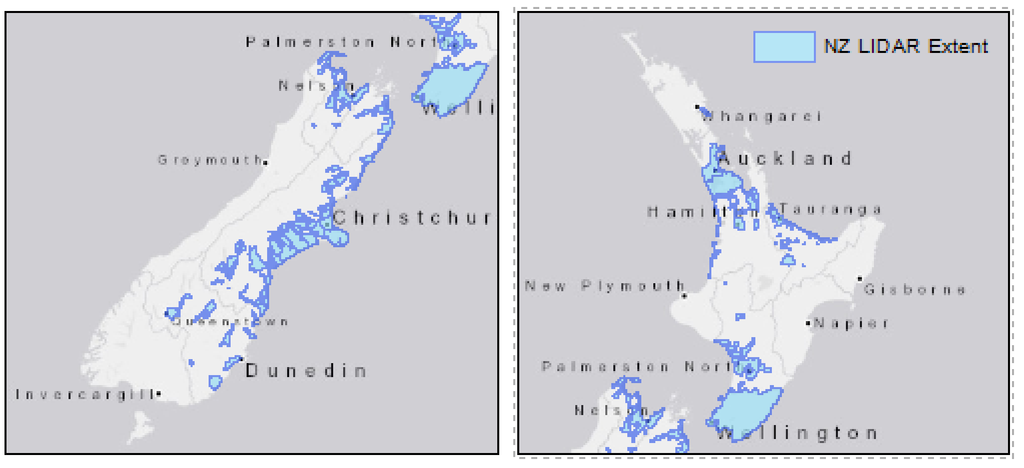
[6] A DEM is considered to represent a 'bare earth' terrain model where the height of built features and vegetation are not included.

from 20 m contour data. Upon comparison and visual inspection in areas where LIDAR overlapped the two DEMs, it became clearer that the NZSoS DEM would be more suitable to fill the areas with no LIDAR coverage.

It was decided to progress with option three. The primary source for the most recently available LIDAR data was LINZ data service and Koordinates Earth's Data Platform. In some cases, individual councils were contacted for data that was not available from these platforms. For example, Northland Regional Council had recently finished a survey that was not yet posted to these platforms. Data collation included sourcing the Digital Surface Model (DSM) LIDAR when available. NZSoS DEM data was sourced from Koordinates.

The extents used from the most recently available LIDAR surveys for New Zealand are shown in Figure 4.19. Approximately 75% of receivers fell within land where LIDAR terrain data was available.

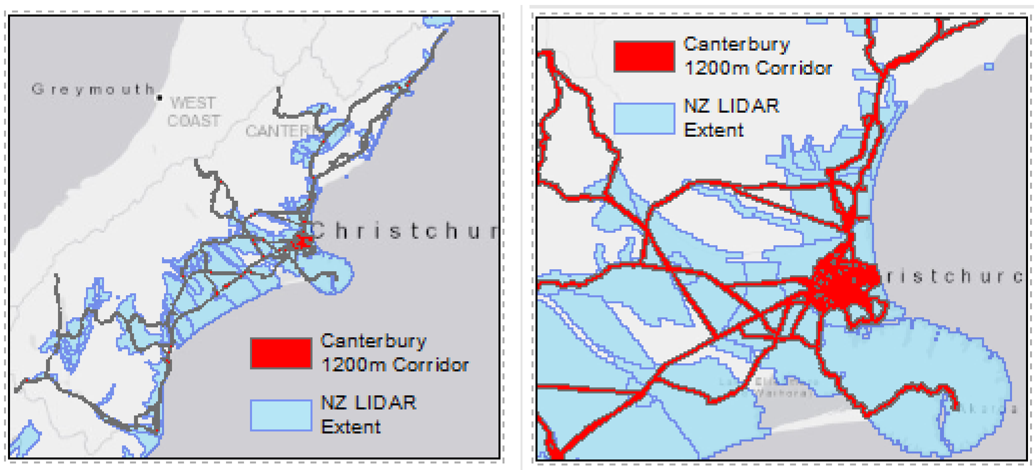
Figure 4.19 LINZ LIDAR Survey extent map for New Zealand



The 1200 m wide national road and rail corridors were split into regional council segments and processed individually to reduce memory and process demands.

Below is an example showing the Canterbury region.

Figure 4.20 Example showing the 1200 m transport corridor for Canterbury and the LIDAR coverage extents



The available regional LIDAR was loaded into Global Mapper and clipped to the Regional 1200 m corridor extent. It was then exported as 5 km DEM tiles ready to be imported into an ArcGIS Mosaic dataset for processing in ArcGIS. The Mosaic footprint capability was used to define an accurate extent of the LIDAR coverage in the region.

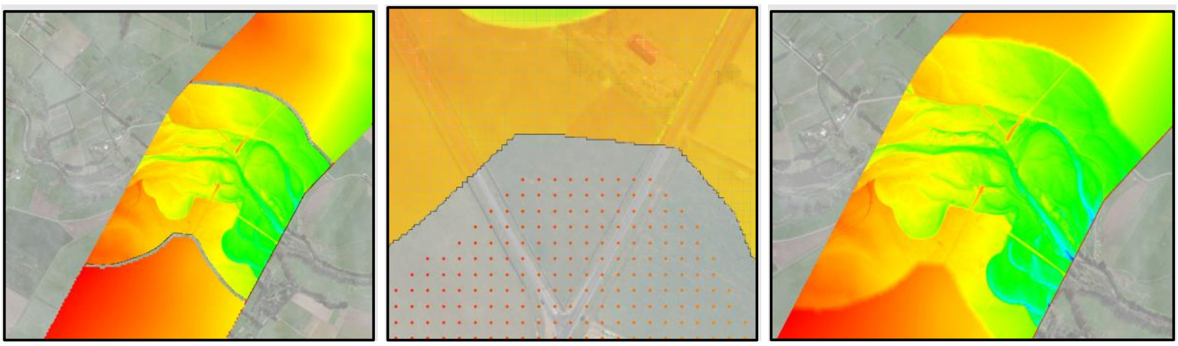
The more accurate LIDAR extent boundary was then buffered by 30 m to generate an interpolation zone between the LIDAR boundary and the lower resolution DEM data. The buffer region was used to erase sections of the 1200 m corridor already covered by LIDAR data (and the 30 m buffer). This left only those regions within the corridor which needed to be filled with the lower resolution DEM data.

The images below show a portion of the corridor illustrating the process.

Figure 4.21 The brown area shows the extent of LIDAR coverage for this section of the corridor. The purple line shows a 30 m buffer region of this extent. This buffer region is then used to erase those areas of LIDAR coverage from the corridor which leaves only those areas requiring data from the lower resolution DEM (NZSoS DEM)

The clipped LIDAR tiles were converted to point data representing the 1 m x 1 m grid. The polygon representing the lower resolution areas was then used to extract and convert the NZSoS DEM to point data representing a 15 m x 15 m grid of points. The result was a set of points representing the combined LIDAR and NZSoS data, with a 30 m gap at the extent boundaries to smooth the transition between the two sets of data.

Figure 4.22 The sequence of images from left to right shows the clipped DEM data from both the LIDAR and the NZSoS DEM, the DEM data converted to point data (1m x1m for LIDAR and 15m x15m for the NZSoS DEM), and the final combined 1m x 1m DEM interpolated from the points



This point data was then used to generate tiled 1 m x 1 m DEM that was suitable for import into the SoundPLAN Model.

The tiles of the combined DEM were mosaicked in ArcGIS to bring together all the tiles into one dataset to represent the combined terrain for the entire 1200 m corridor for the region.

4.2.3.3 Building heights

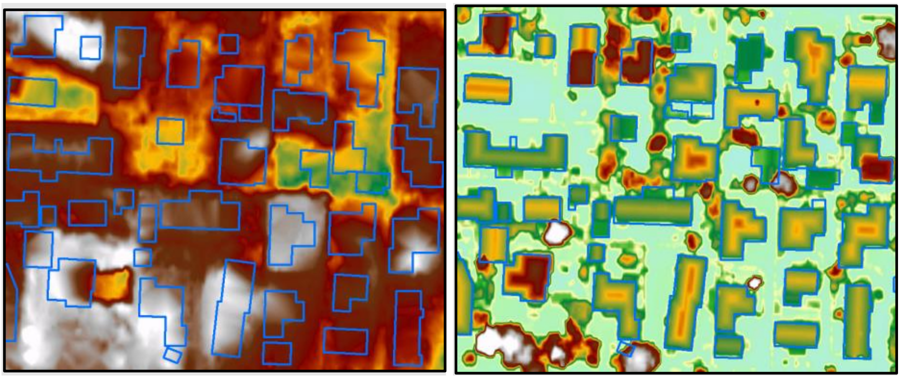
Building heights were based on DEM and DSM data where available in New Zealand. A DEM represents the bare earth terrain only, while a DSM represents the bare ground and all above ground features.

The DEM mosaic was used to interpolate the ground (floor) elevation of buildings. The Zonal Statistics as Table geoprocessing tool was used to extract elevation statistics for each building outline. The tool calculates statistics for all the elevation points contained within each building outline. The median height of points on the DEM that fell within a given building outline was used to record its floor level.

Repeating the process with available DSM data to generate a DSM mosaic provided a means to extract a building's indicative roof elevation which could then be used to calculate a building height. In effect, this meant that a building's height was taken to be the median height from rain gutter to roof ridge.

For areas that did not have DSM coverage a nominal roof height of 6 m was used for residential buildings (including sheds/garages) and 8 m for commercial and industrial buildings.

Figure 4.22 Example showing the difference between a DEM (on right) and a DSM (on left)



4.2.3.4 Road and rail alignments

Roads within the noise model include existing state highways, regional and arterial roads, provided by Abley. Rail alignments for existing operational passenger and freight services were provided by KiwiRail. This includes main trunk lines, secondary main lines, and branch lines as shown in the rail network map spatial dataset[7].

The height of the modelled road and rail centrelines was based on LIDAR data (typically available for populated areas of the country) sourced from the LINZ data service and Koordinates Earth's Data Platform. Processed ground topography from the NZSoS 15 m nationwide DEM was used where there was no LIDAR coverage.

4.2.4 Preparation of bridges

Preparing the data for SoundPLAN required converting the 2D network data to 3D geometries. A DEM was developed that provided terrain information for 1200 m corridors covering the road and rail network. The DEM was used to provide terrain data that could be imported into SoundPLAN and to interpolate the 2D data to 3D.

Modelling of the road and rail in 3D was split into two separate processes to account for bridges. The first process involved overlaying the network data onto the combined DEM and interpolating new 3D datasets for

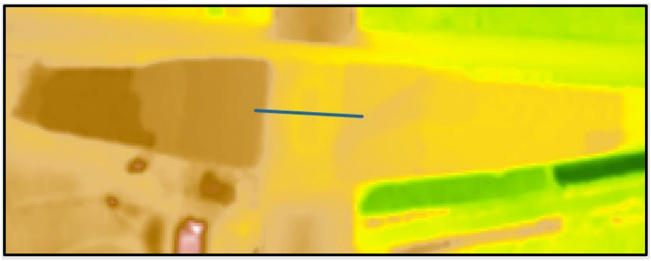
[7] KiwiRail Network Map – Arcgis Hub (https://hub.arcgis.com/datasets/556c4a9c73914fe1983529ddf9ae5099)

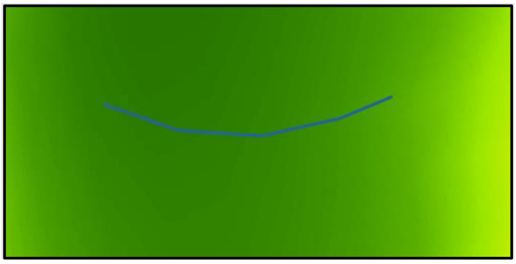
both road and rail. The second process involved separately modelling the 3D bridge data and then merging the generated 3D bridges back into the primary dataset.

The combined DEM represented the bare ground and did not include bridges.

Bridges were modelled based on the start and end points of bridge sections as identified in the LIDAR data. For bridges located in the lower resolution DEM, as can be seen on the right in Figure 4.23, bridge abutment features were not always clear to make out. However, the start and end points were recorded from the bare ground at the start and the end of the bridge.

Figure 4.23 Image on the left showing an example of the abutments of a bridge in the LIDAR DEM but not the bridge itself. The second image shows an example of a bridge spanning in the lower resolution DEM

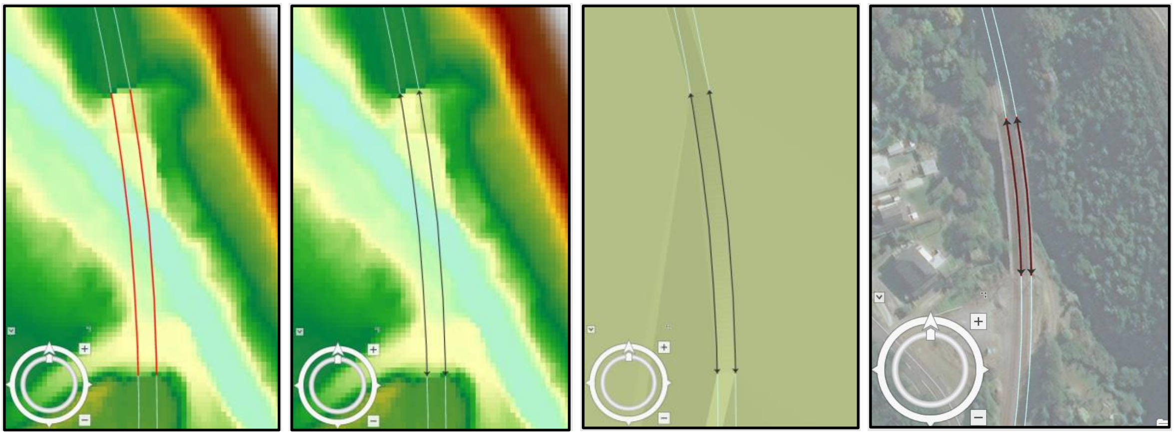




The bridge sections were separated out and dissolved into single features based on Bridge ID. This resulted in single polyline features representing each bridge section. Geometry attributes (start and end of line z values) were added to these separate lines. These values were used to record the start and end height of each bridge feature. This information was used to regenerate each bridge feature into a new 3D feature generated using the start and end elevation values. These lines represented the bridge as a straight line interpolated from the start of the bridge to the end of the bridge. This process did not represent any curvature in the bridge, for example the Auckland Harbour Bridge is recorded as a straight line from start to finish.

Each of these new 3D lines were then copied parallel by 1 m on both sides to provide additional information required for the next stage of the process. The next step involved using this data to generate a Triangular Irregular Network (TIN) surface onto which the original bridge data from the network dataset could be overlaid and interpolated, generating 3D bridge features that contained the attributes required for input into SoundPLAN.

Figure 4.24 Example illustrating the process from left to right. The first image shows the extracted bridge sections, the second shows the start and end points (x,y,z) of the bridge sections. These start and end points are used to generate a 3D feature using the 3D by attribute tool using the start and end points for each line. The third image shows the Triangular Irregular Network (TIN) generated from these 3D features. The original bridge section lines then use the TIN surface to interpolate new 3D geometry for the bridge sections. The last image shows these new 3D lines merged back into the primary dataset



These 3D bridge features were then pasted back into the full network 3D data to complete the network datasets required for SoundPLAN modelling.

After the noise modelling in SoundPLAN was complete, noise levels near bridges were checked to see if there were elevated noise levels at bridge abutments due to steep changes in the gradient of the roads.

A representative sample of approximately 300 bridges across the main urban centres around New Zealand was investigated. The noise contours (Grid noise maps as described in section 4.3.3) produced from the noise modelling were checked for spikes or abnormalities at these bridges. No unexpected spikes in noise levels were observed at any of the bridges checked.

4.2.5 Road traffic noise model

4.2.5.1 Noise modelling software

A 3D representation of the existing environment surrounding each the study alignments was prepared in SoundPLAN (v8.2). The construction of the SoundPLAN road traffic noise model can be separated into the following inputs:

- road traffic data – the traffic volume of each road source line
- pavement surface corrections
- topography – a 3D representation of the existing and proposed earth's surface
- existing noise mounds and alignment (footprint)
- buildings – the location and height of surveyed buildings within the study area
- receptor points – the reference point where the road traffic noise is assessed
- ground absorption – highlighting areas of soft and hard ground
- other model assumptions.

4.2.5.2 Calculation methodology

AECOM completed the road traffic noise mapping using the Calculation of Road Traffic Noise (CRTN) calculation method. The results were presented in $L_{Aeq(24h)}$, a standard method and descriptor for road traffic noise which is in general accordance with New Zealand Standard 6806 and is used extensively in New

Zealand. The CRTN methodology was adjusted for New Zealand road surfaces in accordance with Land Transport New Zealand (LTNZ) Report 326[8] and the NZTA 'Guide to state highway road surface noise[9]'.

The CRTN model was developed based on 18-hour traffic data. However, traffic data was entered as the 24-hour daily traffic (AADT), which resulted in noise levels in the order of +0.2 dB higher than would have been calculated by CRTN based on the 18-hour AADT.

CRTN assumes that traffic is free-flowing, it does not apply to interrupted vehicle flows, such as at intersections, and for low volume roads under 2,000 AADT.

The CRTN calculation method is stated to be accurate to 300 m from the road noise source, however supplementary research carried out by TRL (formally Transport Research Laboratory) found that measured traffic noise levels were in good agreement with predicted noise levels out to a distance of 600 m from a motorway[10].

4.2.5.3 Road surfaces

The road surface finishes were provided by Abley. The road surfaces were prepared as follows:

- A minus 2 dB adjustment was applied for New Zealand conditions in accordance with Transit Research Report 28[11].
- A surface correction relative to asphaltic concrete was made in accordance with LTNZ Research Report 326 and the NZTA 'Guide to state highway road surface noise'.
- The combination of surface corrections for cars and heavy vehicles was made using the equation in the NZTA 'Guide to state highway road surface noise'.
- The combined correction was entered in the modelling software as a total road surface correction applied to the source line.

4.2.5.4 Existing noise barriers

Digitised noise barrier location and dimensions utilised the information within the previous 2019 traffic noise model, which in turn was sourced from the Auckland Noise Improvements Project Business Case. It is noted that data is not available for other NZTA noise barriers.

Existing boundary fences on private properties were not included in the noise model as the condition of these structures was unknown and may not have provided effective acoustic shielding.

All barriers were configured to represent concrete walls – 'hard (fully) reflective' with a reflection loss of 1 dB with a height of 1.6 m unless identified otherwise. Barriers were set to the mean terrain height within SoundPLAN.

4.2.5.5 Bridges/tunnels

Bridges were modelled with the bridge bottom being unreflective, which blocks traffic noise passing under them.

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[8] https://www.nzta.govt.nz/assets/resources/research/reports/326/docs/326.pdf

[9] https://www.nzta.govt.nz/assets/resources/road-surface-noise/docs/nzta-surfaces-noise-guide-v1.0.pdf

[10]https://programmeofficers.co.uk/M27J8/CD/F.9m%20DMRB%2011%20section%203%20part%207%20Traffic%20Noise%20and%20Vibration.pdf

[11] Research Report 28. Traffic noise from uninterrupted traffic flows, Transit, 1994.

For strategic noise mapping, noise from tunnel portals was excluded, which is consistent with the EU good practice guide[12], and the NZTA mapping guidance. Where tunnels were identified, the road noise source stopped at the entrance and resumed at the exit of the tunnel.

4.2.5.6 Ground absorption

The NZTA mapping guidance recommends the use of a ground absorption factor of 1.0, which would infer the presence of soft absorptive ground cover conditions which is generally appropriate for rural areas. In contrast, a ground absorption factor of zero would indicate hard reflective surface cover in the ground plane of the model which is more appropriate for urban areas. Higher absorption in urban areas would result in a higher correction than suitable. Therefore, to account for a mixture of hard and soft surfaces within cities and urban areas, a factor of 0.6 would be more appropriate.

Ground absorption factors were adjusted to factor in ground type appropriate for urban and rural areas. Urban areas for high residential areas such as Auckland and Wellington were set at 0.6 and rural areas were set at 1.0.

4.2.6 Rail noise model

4.2.6.1 Calculation methodology

New Zealand does not currently have a standard or guidelines relevant for rail noise mapping or control. It was considered the UK Calculation of Rail Noise (CRN) 1995 method would be appropriate for undertaking the noise mapping due to the similarity to the Calculation of Road Traffic Noise (CRTN).

This calculation method produces results in terms of L_{Aeq} noise levels which aligns with the road traffic noise model.

4.2.6.2 Data preparation

The rail data was prepared by matching rail linear reference points to matching chainage data (mile posts) linked to Excel look-up documents. Information for the rail network, mile posts, and associated look-up tables were provided by KiwiRail. The table information contained start and end measures associated with the chainages of each line. An example is shown in Figure 4.25.

Figure 4.25 Example of tabular look-up data based on start and end measure

KR_LINE	CLASSIFICATIONID	ASSETNUM	STARTFEATURELABEL	STARTOFFSET	STARTMEASURE	ENDFEATURE	ENDOFFSET	ENDMEASURE	LINESPEED	OPSPEED	CURVESPEED
NIMT	MAINL	1000000	0	0	0	0	542	542	100	20	
NIMT	MAINL	1000000	0	542	542	0	632	632	100	20	60
NIMT	MAINL	1000000	0	632	632	0	811	811	100	60	60
NIMT	MAINL	1000000	0	811	811	1	61	1058	100	60	
NIMT	MAINL	1000000	1	61	1058	1	285	1282	100	60	60
NIMT	MAINL	1000000	1	285	1282	1	359	1356	100	60	
NIMT	MAINL	1000000	1	359	1356	1	571	1568	100	60	60
NIMT	MAINL	1000000	1	571	1568	2	16	2012	100	60	

The process used the Linear Referencing tools in ArcGIS to build a linear referenced route and events for each of the relevant input tables. The tools provided a way to add a measure to a line that could be used as reference to locate the start and end measures associated with the look-up table. The process was repeated for each input and then consolidated to generate a rail network spatial dataset containing the attribute data required by SoundPLAN.

[12] European Commission Working Group Assessment of Exposure to Noise, '*Good practice guide for strategic noise mapping and the production of associated data on noise exposure*', December 2003. http://ec.europa.eu/environment/noise/pdf/wg_aen.pdf

The accuracy of linear referencing relied on the accuracy of the incoming information, specifically the chainage point locations and the accuracy/validity of the associated tabular data. Once mapped it became apparent that in some situations the tabular recorded chainage location was inconsistent with the physical location. This could be seen when a linear referenced bridge structure would not exactly line up with physical location.

This was particularly noticeable with the curve radius information held in the tabular data. When mapping the curve radius data, it was found that the curve values did not always match the physical location of the curve. This was fixed by spatially identifying which sections of rail required each radius correction, then applying the corrections directly in SoundPLAN in the rail strings. This process is discussed in further detail in section 4.2.6.7.

4.2.6.3 Train volumes

Train volume data was provided by KiwiRail to calculate daily train movement volumes for each of the day/evening/night-time slices for the L_{den} and L_{night} noise metrics. This allowed the calculation of the L_{den} noise level indicator required for the human and economic assessments. The train volumes provided were for the year 2019.

Rail noise data, both for passenger and freight movements between Masterton and Woodville was not collated through the data collection exercise, meaning that noise levels for this area of the Northern Wairapapa Line are not reflected in the modelling work undertaken. At the time of reporting, it is understood that there are no regular passenger or freight services between Masterton and Woodville on the Northern Wairapapa Line, with only occasional freight services operating as required.

4.2.6.4 Train speeds

Posted train speed for each corridor and section was sourced from KiwiRail and applied along the rail centreline within the model.

Where there is a speed change along the corridor (for example from 100 km/h to 80 km/h), it was assumed there would be no gradual speed change (deceleration or acceleration) between the two areas for consistency throughout the network.

The model assumed trains would travel at a constant speed through any stations or stops or yards. As trains will not be stopping anywhere within the network, predictions around stations or yards may be elevated, giving a more conservative noise level.

4.2.6.5 Train types, measurements, and source levels

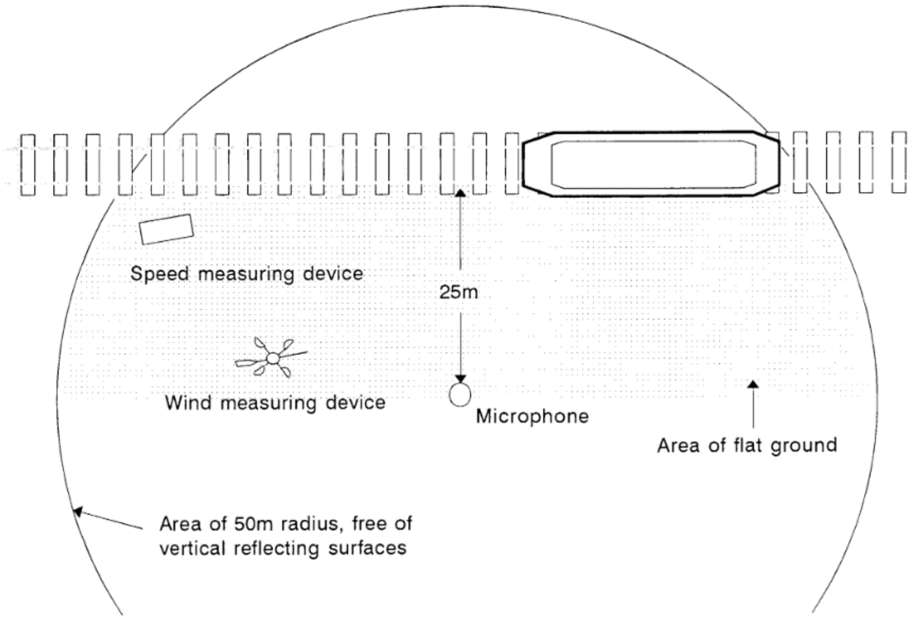
Specific locomotive types were provided by KiwiRail with the train volumes data. Source noise measurements of each type of locomotive were carried out as per the following section. The source noise level data was then entered into the SoundPLAN library and used in the modelling.

New Zealand rolling stock measurements were completed in accordance with CRN and ISO 3095 to obtain the Sound Exposure Level (SEL) for relevant locomotives, wagons, and passenger trains. Measurements were carried out because the CRN method provides noise data based on rolling stock in the UK, so measurements were required for rolling stock in New Zealand to account for any differences.

Some train measurements could not be undertaken due to closure of some tracks in the South Island, COVID-19 restrictions, and the running of different train types than anticipated on measurement days.

Measurements were conducted in general accordance with CRN requirements as shown in Figure 4.26 below, with the addition of another sound level meter positioned at 7.5 m from the track at a height of between 1.5 and 3.5 m. A video recording device was set up alongside for reference to number of carriages and locomotive type.

Figure 4.26 Site layout for the measurement of Sound Exposure Levels (SELs) from individual rail vehicles



The microphone that was set up at 25 m from the track recorded SEL and the microphone set up at 7.5 m from the track recorded continuous noise levels using the L_{Aeq} descriptor. The L_{Aeq} data was used to determine the time stamp of the vehicle of interest to determine the SEL as per ISO 3095.

Where multiple locomotives or wagons of the same type were recorded, the average of measurements was used to determine a correction from the baseline.

A separate SoundPLAN model of the measurement locations was created to calibrate the calculated corrections and ensure modelled noise levels of each train type accurately reflected measurements.

The results of the measurements and corrections obtained in SoundPLAN are summarised in Table 4.3, Table 4.4 and

Table 4.5.

Table 4.3 Summary of measurements, processing, and assumptions for locomotives

Type	Measured speed (km/h)	Measured SEL	SoundPLAN level	SoundPLAN correction based on calibration from measurement	Train category	Comment
DL Class Diesel Loco	40	86 dBA	37 dB L_{Aeq24h}, 86 dBA SEL	−4	7: Locomotives (Diesel)	
DF Class Diesel Loco	45	88 dBA	39 dB L_{Aeq24h}, 88 dBA SEL	−2	7: Locomotives (Diesel)	
DC Class Diesel Loco	N/A	N/A	Using DF Loco as similar spec	−2	7: Locomotives (Diesel)	Used DF Loco as similar spec
Shunt Loco Class	N/A	N/A	Using DF Loco as similar spec	−2	7: Locomotives (Diesel)	Used DF Loco as similar spec
DX Class Diesel Loco	N/A	N/A	Using DF Loco as similar spec	−2	7: Locomotives (Diesel)	Used DF Loco as similar spec

Table 4.4 Summary of measurements, processing, and assumptions for wagons

Type	Measured speed (km/h)	Measured SEL	SoundPLAN level	SoundPLAN correction	Train category	Comment
Wagon 60 CFT	40	72 dBA	23 dB L_{Aeq24h}, 72 dBA SEL	15	12: 2007 Wagons	
Wagon 50 CFT	40	71 dBA	22 dB L_{Aeq24h}, 71 dBA SEL	14	12: 2007 Wagons	
Wagon 40 CFT	N/A	N/A	Using Wagon 50 CFT	14	12: 2007 Wagons	Used Wagon 50 CFT
Wagon Box	40	73 dBA	23.7 dB L_{Aeq24h}, 73 dBA SEL	16	12: 2007 Wagons	
Wagon Log	45	76 dBA	26 dB L_{Aeq24h}, 76 dBA SEL	18	12: 2007 Wagons	
Wagon Hopper	N/A	N/A	Using avg	16	12: 2007 Wagons	Used Box Wagon
Service Wagon	N/A	N/A	Using Log Wagon	18	12: 2007 Wagons	Used Log Wagon

Table 4.5 Summary of measurements, processing, and assumptions for passenger and electric trains

Type	Measured speed (km/h)	Measured SEL	SoundPLAN level	SoundPLAN correction	Train category	Comment
Wellington Passenger	70	78 dBA	29 dB L_{Aeq24h}, 78 dBA SEL	16	11: 2007 EMU	
Auckland EMU	N/A	N/A	Using Wellington Passenger as similar spec	16	11: 2007 EMU	Used Wellington Passenger as similar spec
Passenger Wagon Pukekohe	40	71 dBA	22 dB L_{Aeq24h}, 71 dBA SEL	15	12: 2007 Wagons	Diesel Train w Passenger Wagons
Electric EF	N/A	N/A	Basing off locomotive corrections as no data	−2	8: Diesel Locomotives	Based on locomotive corrections as no data

4.2.6.6 Track corrections

CRN allows for corrections to account for the additional noise produced by crossings (where tracks overlap), condition of track form, ballast type, and radius of track which were applied. This data was provided as attributes in the rail strings provided by KiwiRail and was screened by spot-checks of the data in some areas.

Attribute descriptions as set out in Figure 4.18 identify the pre-defined corrections as per CRN method presented in SoundPLAN. These fields were populated within GIS for import into the SoundPLAN model to reduce processing time.

Corrections for track conditions as per CRN were applied as follows:

- continuously welded rail (CWR) with concrete/timbers + ballast (+0 dB)
- jointed track (18.3 m) length, points and crossing (+2.5 dB)
- slab track (+2 dB).

Where rail tracks cross a crossing/turnout, a correction of +5 dB was applied as per the agreed methodology.

4.2.6.7 Curve corrections

Corrections for track radius were applied as per the following:

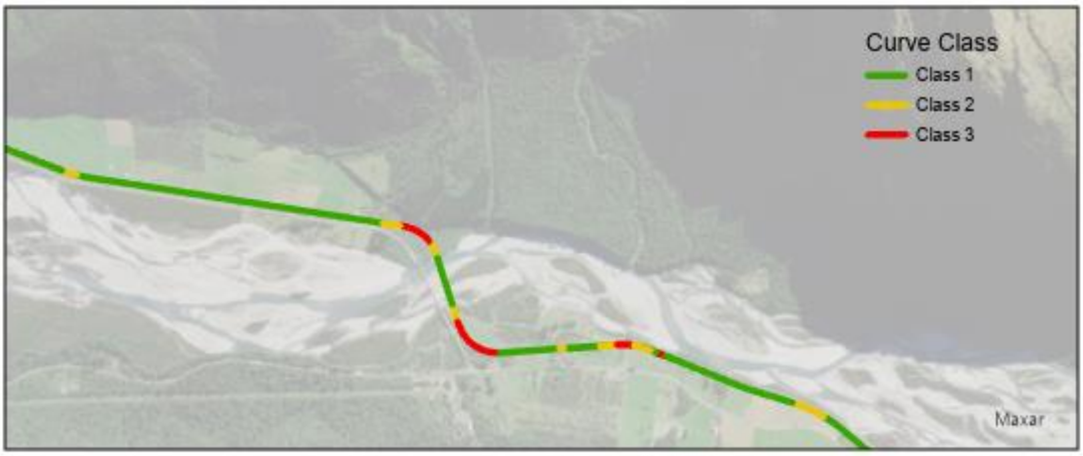
- R>500 m (+0 dB – Curve Class Value 1)
- 300 m<R<500 m (+3 dB – Curve Class Value 2)
- R<300 m (+8 dB – Curve Class Value 3).

Corrections were applied at the start of a radius change for the whole length of the different attributes and stopped where the string would resume in a straight line. For crossings and bridges, corrections were applied at the start of the bridge/crossing and stopped at the end of the bridge/crossing.

On inspection of the tabular data and discussions with KiwiRail, it was identified that the problem with mismatching of tabular to spatial data was occurring because the tabular chainage information for the start and end of curve was not matching the physical location data from the chainage post data. After a discussion with KiwiRail, it was confirmed that data provided was the most accurate available. An alternative approach was therefore required to map the correct locations and magnitudes of the curves.

The solution chosen was to calculate the curve radius using the geometry of the rail lines. The network was broken up into 100 m segments, the start and end of line bearing was calculated, and then the known 100 m length and bearing difference information was used to calculate the curve radius for each segment. These calculated curve radii were then transferred back to the original rail data and used to calculate the Curve Class Value for SoundPLAN modelling. Curve radii less than 300 were classed as Class 3, less than 500 and greater and equal to 300 were classed as Class 2, and greater than or equal to 500 were classed as Class 1.

Figure 4.27 Example showing the results of the Curve Class calculations



4.2.6.8 Noise barriers

There are no noise barriers included in the rail noise model as there was no dataset of noise barriers along the rail corridor available.

4.2.6.9 Bridges/tunnels corrections

The bridges were configured to be 'self-screening'. Bridge corrections were applied as follows:

- no bridge (+0 dB)
- concrete bridges and viaducts with parapets (+1 dB)
- steel bridges with parapets (+4 dB)

- steel bridges; box girder (gr) with rails fitted to gr + orthoptic slab rail bearing (+9 dB).

For strategic noise mapping, noise from tunnel portals were excluded, which is consistent with the EU good practice guide[13]. Where tunnels were identified, the rail noise source stopped at the entrance and resumed at the exit of the tunnel.

4.2.6.10 Ground absorption

In line with the road model, ground absorption factor in urban environments was set at 0.6 and rural environments set at 1.0.

4.3 Noise modelling

4.3.1 Management of large datasets

Two master models split between the North Island and the South Island of New Zealand were set up within SoundPLAN before using the built in Tiles function to split the country further into field size 2 x 2 km assessment grids. Using the SoundPLAN Tiles tool to model in smaller sections allowed for increased efficiency in modelling and data management.

4.3.2 Façade noise maps (FNM)

Noise levels were calculated at the centre of each façade, 1.5 m above each floor height, 1 m from the façade. The highest modelled noise level for each dwelling was taken. Noise at all façades within 600 m of the road or rail centreline was modelled. Free-field equivalent noise levels incident on the buildings were obtained by applying a correction of −2.5 dB to façade noise levels to account for façade reflections.

Utilising Tiles eliminated the problem previously encountered in the 2019 model where post processing led to contours not overlapping correctly, as surrounding propagations are also taken into consideration by the model itself. It also allowed for easy distribution of the same model over multiple modelling computers.

4.3.3 Grid noise maps (GNM)

Grid noise map (GNM) calculations were used to visualise noise from the transport networks. The noise contours produced are shown in the online GIS web map. Noise contours were calculated at a 10 m x 10 m grid size at a receiver height of 1.5 m above local ground level.

4.3.4 Conversion to noise exposure levels

The assessment of population noise exposure was undertaken in consideration of the following noise level indicators in line with the human health and social cost requirements and New Zealand standards:

- $L_{Aeq\,(24h)}$
- L_{den} and L_{night}.

The CRTN algorithm calculates results in $L_{A10(18h)}$. To convert these results to $L_{Aeq(24h)}$ a minus 3 dB adjustment was made. This adjustment was implemented in the software.

[13] European Commission Working Group Assessment of Exposure to Noise, '*Good practice guide for strategic noise mapping and the production of associated data on noise exposure*', December 2003. http://ec.europa.eu/environment/noise/pdf/wg_aen.pdf

The 'Environmental Noise Guidelines for the European Region' (dated 2018)[14] guideline value of 53 dB L_{den} for road traffic noise would approximately correlate with a noise level of 50 dB $L_{Aeq\ (24h)}$, while the European guideline value of 45 dB L_{night} would approximately correlate with a noise level of 51 dB $L_{Aeq\ (24h)}$. These conversions are based on the non-motorway conversion Method 3, from the TRL report, 'Method for converting the UK road traffic noise index $L_{A10(18h)}$ to the EU noise indices for road noise mapping', 2006[15]. These adjustments for L_{den} and L_{night} were applied during the post-model processing stage.

The L_{den} for rail noise was calculated within SoundPLAN using the rail volumes provided by KiwiRail.

4.3.5 Model limitations

4.3.5.1 Road

The following limitations applied to road noise modelling:

- New Zealand-specific research is currently underway with draft findings indicating that the uncertainty inherent with applying CRTN within New Zealand is in the region of +/− 5 dB. This is discussed further in section 4.5.2.
- Noise levels were predicted beyond the standard 300 m range for validity specified in the CRTN standard.
- Noise monitoring was not undertaken to validate the predicted road traffic noise levels.
- The incidence and consequent noise from truck engine brakes or from single noise events (eg, a loud motorbike without a muffler) was not included within the noise model.
- Posted speed limits were used in the noise modelling. The speed limits were not adjusted for intersections.

4.3.5.2 Rail

The following limitations applied to rail noise modelling:

- Noise from track sidings, rail stabling yards, or other types of supporting or maintenance infrastructure were not included within the model.
- Acceleration and deceleration around train stations was not modelled.
- Trains were assumed to travel at sign posted speed and were not modelled to account for the possible variation in speed to account for track conditions or rail gradients.
- A limited number of train types were measured, therefore some of the source values used were based on calculation/assumption as per section 4.2.6.
- Train noise source levels were based on measurements undertaken by AECOM. Source levels were developed for each train type. However, corrections for individual notch settings were not defined or modelled.
- Noise from the use of Klaxon horns or other safety devices (eg, warning bells or PA systems) were not modelled.
- Whilst CRN factors corrections for crossings (where tracks overlap), condition of track form, ballast type, and radius of track, this calculation method is not intended for calculation of all aspects of railway noise and does not take into account the potential effect of variability of rail head roughness. Limitations of CRN are provided in the literature study.

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[14] *Environmental Noise Guidelines for the European Region* (2018), http://www.euro.who.int/en/publications/abstracts/environmental-noise-guidelines-for-the-european-region-2018

[15] For DEFRA by TRL and Casella, *Method for converting the UK road traffic noise index $L_{A10(18h)}$ to the EU noise indices for road noise mapping*, 2006, https://pdfs.semanticscholar.org/32b4/09d29b0d811f0c36afe4e01529beea802caa.pdf.

- Private rail networks were excluded from the noise model.
- Rail corridors without rail volume data were excluded from the assessment.

4.4 Final outputs

4.4.1 Results for cost model

Following the process set out in the earlier sections of this report, noise levels were predicted at each dwelling. The predicted noise levels were then joined to the dataset of dwellings.

A summary of the count of the population and dwellings per health effect for each year is presented in Table 4.6 for information. Onset of health effects is determined to be above certain noise thresholds as summarised in the headings in Table 4.6 and as discussed in sections 3.3.6, 3.4.5, and 3.7.5.

It is noted that these counts were further processed in the cost model to account for relevant factors such as the percentage of the adult population and the percentage of people more or less annoyed by noise, which is why the counts shown in section 5.6 are lower overall than those shown here.

The full dataset of the count of affected dwellings and population (as of 2018) per 1 dB noise increment per district, region, and health district was used in the cost model as per section 5.2.2.

Table 4.6 Summary of counts of affected dwellings and population

Section	Health issues					
	Sleep disturbance (L_{night} >= 33 dBA Rail, >= 43 dBA Road)		Annoyance (L_{den} >=34 dBA Rail, >= 40 dBA Road)		Ischaemic heart disease (L_{den} >= 53 dBA Road and Rail)	
	Count of dwellings	Count of people	Count of dwellings	Count of people	Count of dwellings	Count of people
North Island Road	189,769	649,884	628,295	2,059,985	170,622	584,104
South Island Road	119,193	379,774	237,917	655,472	61,319	169,537
North Island Rail	150,504	483,927	153,444	491,972	43,933	140,036
South Island Rail	63,730	169,453	62,411	165,958	15,168	39,473

4.4.2 Presentation of road modelling results

The NZTA mapping guidance recommends the graphical presentation of model results in the format shown in Table 4.7 and Table 4.8. The following table shows the range of values included in the individually coloured solid filled contours. Contour lines are shown on the maps at 1 dB intervals. The contours were simplified by reducing the number of points used to reduce file size. These noise contours are shown on the SCON web map as described in section 4.4.4.

Table 4.7 Road noise contour colouring

Interval	Noise zone fill (pastel colour)
50 dB ≤ $L_{Aeq(24\ hour)}$ < 55 dB	Light blue
55 dB ≤ $L_{Aeq(24\ hour)}$ < 60 dB	Light green
60 dB ≤ $L_{Aeq(24\ hour)}$ < 65 dB	Yellow
65 dB ≤ $L_{Aeq(24\ hour)}$ < 70 dB	Orange
$L_{Aeq(24\ hour)}$ ≥ 70 dB	Pink

The following table shows the recommended colouring for sensitive receivers identified as Protected Premises and Facilities (PPF), classified according to New Zealand Standard 6806. The colouring of PPFs indicates the noise category in which each dwelling is located.

Table 4.8 Protected Premises and Facilities colouring

Category	Interval	Residential outline (solid colour)
A	$L_{Aeq(24\ hour)}$ < 64 dB	Green
B	64 dB ≤ $L_{Aeq(24\ hour)}$ < 67 dB	Orange
C	67 dB ≤ $L_{Aeq(24\ hour)}$	Red

4.4.3 Presentation of rail modelling results

There is currently no KiwiRail guidance to noise mapping for rail. Based on previous AECOM experience, 5 dB solid filled noise contours as data layers showing the modelled $L_{Aeq(24h)}$ noise levels have been presented on the SCON web map as described in section 4.4.4. The contours were simplified by reducing the number of points used to reduce file size. The following ranges and colour coding were used:

Table 4.9 Rail contours interval colourings

Interval	Noise zone fill (pastel colour)
50 dB ≤ $L_{Aeq(24\ hour)}$ < 55 dB	Light blue
55 dB ≤ $L_{Aeq(24\ hour)}$ < 60 dB	Light green
60 dB ≤ $L_{Aeq(24\ hour)}$ < 65 dB	Yellow
65 dB ≤ $L_{Aeq(24\ hour)}$ < 70 dB	Orange
$L_{Aeq(24\ hour)}$ ≥ 70 dB	Pink

4.4.4 GIS outputs

The SCON dashboard was prepared to allow users to interact with the outputs of the cost model (as detailed in section 5) through an interactive map and charts. Further detail around the SCON dashboard is provided in Appendix B.

A separate web map has also been prepared with a focus on displaying the predicted noise contours and PPF maps with colour coding, along with functionality to print PDF maps. The web map also allows the user to view the outputs of the cost model.

4.5 Uncertainty

4.5.1 Sources of uncertainty

4.5.1.1 Error due to the modelling algorithm

It is understood that further New Zealand-specific research is currently underway with draft findings indicating that the uncertainty inherent with applying CRTN to New Zealand conditions is of a similar order to that found in the 2004 Queensland study (+/− 4 or 5 dB).

This uncertainty is largely due to the specific conditions found in the southern hemisphere. This is because, in all of the examples cited above, a region-specific correction had to be incorporated into the CRTN model in order to better reflect local conditions. It is thought that this residual difference could be due to the type and percentage of heavy vehicles observed (ie, source inputs) (Peng et al., 2017) as well as the absorption provided by the terrain (ie, the propagation model).

Based on the above it is considered that +/− 5 dB is a reasonable estimate of the uncertainty related to the modelling algorithm.

4.5.1.2 Error due to chosen assumptions and quality of inputs

While care was taken to ensure that all modelling inputs were processed and imported into the noise models correctly, it was likely that mistakes or discrepancies in the input data could have affected the results. There would have also been uncertainty introduced to the results based on the assumptions used in the modelling exercise.

Examples where errors in the input data may have come about include errors due to:

- the spatial position of input data
- uncertainty in rail noise data deduced from measurements
- the location of receiver points along façades, particularly where bedrooms are set back or face away from transport noise sources
- façade correction used – assumed 2.5 dB correction however the actual value in reality would depend on a number of factors, including frequency of the source noise, angle of view, façade width, distance from the noise source, building orientation etc.

It is noted that these deviations would likely lead to over-predictions of noise (and therefore cost) rather than under-predictions.

4.5.2 Estimation of uncertainty from noise modelling

The sources of uncertainty listed above can largely be classed as introducing two categories of error – random error and systematic error.

Random error is error that affects the precision of the measured value around the true value. Random error in this research would include the error due to the modelling algorithm, error due to uncertainty in the modelled locations of receivers, and error due to the façade correction used, among other sources.

For this research, noise levels were modelled for a very large number of receivers across the country, therefore it is considered that any random error will have an insignificant effect on the final results on the social cost of noise determined in this research, as any random error would have effectively been averaged out when the number of receivers per 1 dB noise interval were aggregated (as set out in section 5).

Systematic error is error that is replicated across all measurements in a given dataset. For this research, sources of systematic error could include error inherent to the input data, for example spatial position (if this was consistent across the models), error from the rail source noise measurements, and any other bias in the modelling process, among other sources.

Systematic error introduced in the modelling process would be reflected in the outcomes of this research and is therefore the basis of our overall estimation of uncertainty introduced in the noise modelling process.

While it would be difficult to determine all sources of uncertainty/error, it is considered that an overall estimated uncertainty value of +/− 2 dB (95% uncertainty confidence) is reasonable for this study. This value takes into consideration the realistic effect of the most likely sources of systematic error (error from the input data and measurements), while avoiding overestimation of uncertainty.

Regardless of the uncertainty introduced during the noise modelling exercise, it is noted that the uncertainty introduced in the cost modelling is much greater than any uncertainty introduced in the noise modelling. This is shown in the results of the sensitivity analysis set out in section 5.6.1, where checking the high and low scenario disability weighting and QALY values produce final cost values +383% and −48% about the central scenario value for road traffic noise, and +292% and −47% for rail noise.

4.6 Conclusion

Noise modelling of the road and rail network across New Zealand was completed. The noise modelling was carried out separately between road and rail, and the North and South Island, for a total of four noise models.

The modelling required identification of dwellings and population counts per dwelling. This was done through an automated process by applying a number of classification rules based on the average population per statistical area, the number of building outlines per parcel, and the number of address points per parcel.

Topography was processed using a combination of the terrain information available per region. Bridges were processed according to abutments identified on the DEM. Other inputs in the noise model included buildings and the road/rail alignments.

The noise models were managed in SoundPLAN 8.2 using the tiling function, allowing the model to be run in discrete 2 km x 2 km sections. Façade noise maps and grid noise maps were calculated from each of the noise models. The façade noise map results were used to determine the counts of people and dwellings that fell within each 1 dB noise band, which was then entered into the cost model. The grid noise maps were entered into the online GIS web map to be visualised on the platform.

5 Cost modelling

5.1 Overview

A detailed cost model has been developed to monetise the social impacts associated with noise. The model has been developed based on assumptions and parameter values identified in the acoustic modelling and literature review undertaken as part of the broader project, as well as New Zealand Treasury modelling guidance. The framework for the model is provided in Appendix A3.

The cost model seeks to quantify the social costs associated with three key health issues which were identified through the literature review, these being:

- annoyance
- sleep disturbance
- ischaemic heart disease (IHD).

While the cost model is focused on these impacts, it is acknowledged that there are other health issues and social impacts associated which were identified in the literature review but have not been modelled due to insufficient evidence or research to support the quantification of those impacts.

Results generated by the model were estimated at 1 dB increments and disaggregated by geographic unit at various levels (ie, territorial authority, regional council, and health district) for the three health endpoints. Modelled results were used to estimate:

- the number of residents who suffer from one or more of the conditions at each noise interval
- each condition's economic and societal cost as a whole at each noise interval.

Table 5.1 below provides a summary of the key modelling outputs which have been derived from the social cost model, while the assumptions and parameter values used in the cost model are summarised in section 5.2.

Table 5.1 Key modelling outputs

Health endpoint	Annoyance	Sleep Disturbance	Ischaemic heart disease (IHD)
Modelling outputs	Number of residents (adult population) highly annoyed at each noise interval $ health value of highly annoyed residents at each noise interval	Number of residents (adult population) with highly disturbed sleep at each noise interval $ health value of highly sleep disturbed residents at each noise interval	$ health value for residents (35+ population) with IHD at each noise interval

5.2 Modelling assumptions and values

5.2.1 Cost model assumptions

The general parameters used in the cost model are set out in Table 5.2. General parameters are those inputs or assumptions which underpin the cost model and its outputs.

Table 5.2 General parameters

Parameter	Value
Price year (New Zealand)	FY2022
Appraisal period	1 year
New Zealand population (March 2022) (Stats NZ, 2022)	5,127,200

Although the input data for the noise model came from earlier years, the cost model used the financial year ending 30 June 2022 as the base year. Using nominal gross domestic product (GDP) data (New Zealand Treasury, 2021), all monetary values used in this assessment were adjusted to the base year.

5.2.2 Acoustic modelling

Population noise exposure inputs utilised in the cost model were generated from the acoustic model. The acoustic model combined population data and noise modelling to estimate the number of people (buildings) exposed to road and rail transport noise at increments in 1 dB. For further detail on the acoustic modelling methodology, refer to section 4 of this report.

Census data from 2022 was utilised to apportion population by territorial authority, regional council, and health district.

The key outputs from acoustic modelling that were used in the cost model are detailed below in Table 5.3.

Table 5.3 Acoustic model key outputs

Parameter	Description
L_{den}	Day-evening-night weighted sound pressure level (dB) weighted over a 24-hour period that includes a penalty of the evening and night-time level to account for the elevated noise sensitivity in these time periods
L_{night}	Average sound pressure for an 8-hour night-time period (23:00-07:00)
Residential building noise exposure	Residential buildings exposed to transport noise at 1 dB increments
Residential building occupancy	Occupancy (number of people) of each residential building

5.2.3 Exposure-response relationship assumptions

Exposure-response relationships were sourced from the literature review. In addition, the literature review identified and critiqued existing research relating to exposure-response relationships, including studies that are specifically relevant to New Zealand.

The exposure-response formulas and relative risk ratios derived from the literature review are summarised in this section. In addition, detailed analysis and commentary surrounding the formulas used in the cost model are provided in the literature review in Appendix A3.

5.2.3.1 Sleep disturbance

The literature review recommended that the relationship between dB and the percentage of population highly sleep disturbed (HSD) established by the WHO review (Basner & McGuire, 2018) be used to assess road and rail transport noise.

The WHO (2009, 2011, 2018) identified an exposure-response relationship that relates the percentage of residents sleep disturbed (SD) and HSD to L_{night} (ie, average noise levels during night-time, as measured outdoors). The WHO review (2018) concluded that the key outcome of %HSD was considered most appropriate for determining actions and results concerning transport noise.

The recommended relationship between L_{night} and %HSD is a summary of the meta-analysis undertaken by Basner and McGuire (2018). The relationship is represented by a quadratic regression function which illustrates the 'line of best fit' derived from the scatterplot of results of different studies within the meta-analysis. The general formula for a quadratic equation is:

$$y = a - b * x + c * x^2.$$

The analysis considered the exposure-response relationship between noise and HSD, which is described in Equation 1 and Table 5.4 below:

Equation 1 Dose-response relationship for sleep disturbance

$$\%\ Highly\ Sleep\ Disturbed\ (\%\ HSD) = \ a - b * L_{night} + c * {L_{night}}^2$$

Table 5.4 Dose-response relationship for sleep disturbance

Health endpoint	Noise source	Noise metric	Coefficient values	Dose-response relationship	Range of applicability (decibels)	Source
Sleep disturbance – (%HSD) (Adult population only)	Road traffic noise	dB L_{night}	a = 19.7885 b = 0.9336 c = 0.0126	%HSD = 19.7885–0.9336 × L_{night} + 0.0126 × L_{night}^2	Low = 43 High = 65	Basner & McGuire (2018)
	Railway noise	dB L_{night}	a = 67.5406 b = 3.1852 c = 0.0391	%HSD = 67.5406–3.1852 × L_{night} + 0.0391 × L_{night}^2	Low = 33 High = 65	Basner & McGuire (2018)

5.2.3.2 Annoyance

The literature review recommended that the relationships between dB and the percentage of the population highly annoyed (HA) established by WHO (Guski et al., 2017) be used in this assessment for both road and rail transport noise.

The recommended relationship between L_{den} and %HA is a summary of the meta-analysis undertaken by Guski et al. (2017). The relationship is represented by a quadratic regression function which illustrates the 'line of best fit' derived from the scatterplot of results of different studies within the meta-analysis.

This analysis considered the exposure-response relationship between noise and HA, as described in Equation 2 and Table 5.5 below:

Equation 2 Dose-response relationship for annoyance

$$\%\ Highly\ Annoyed\ (\%HA) = \ a - b * L_{den} + \ c * {L_{den}}^2$$

Table 5.5 Dose-response relationship for annoyance

Health endpoint	Noise source	Noise metric	Coefficient values	Dose-response relationship	Range of applicability (decibels)	Source
Annoyance – % Highly Annoyed (%HA) (Adult population only)	Road traffic noise	dB L_{den}	a = 78.927 b = 3.1162 c = 0.0342	%HA = 78.9270–3.1162 × L_{den} + 0.0342 × L_{den}^2 %HA = 78.9270–3.1162 × L_{den} + 0.0342 × L_{den}^2	Low = 40 High = 75	Guski et al. (2017)
	Railway noise	dB L_{den}	a = 38.1596 b = 2.05538 c = 0.0285	%HA = 38.1596–2.05538 × L_{den} + 0.0285 × L_{den}^2	Low = 34 High = 75	Guski et al. 2017

5.2.3.3 Ischaemic heart disease

Assessing noise impacts on IHD, the literature review recommended that the relative risk relationship provided by Vienneau et al. (2020) be adopted. Vienneau et al. (2020) identified lower risks but a slightly higher quality of evidence for railway noise and a narrower 95% confidence interval for road traffic noise than the Kempen et al. (2018) meta-analysis.

Quantification of this impact has been estimated through consideration of the relative risk relationship of IHD and transport noise (per 10 dB increase) as described in Table 5.6 below:

Table 5.6 Relative risk ratio of ischaemic heart disease (IHD)

Health endpoint	Noise source	Noise metric	Relative Risk per 10 dB increase [95% confidence interval]	Range of applicability (decibels)	Source
IHD Incidence (Adult population 35+ years only)	Road traffic noise	dB L_{den}	RR = 1.08 [1.01 – 1.15]	Low = 53	Vienneau et al. (2019)
	Railway noise	dB L_{den}	RR = 1.01 [0.99 – 1.03]	Low = 53	Vienneau et al. (2019)

Note that the rural exposure-response function was conservatively applied in the modelling.

In the absence of a regression equation, the additional risk of IHD due to transport noise exposure (dB) metric was used as a proxy to define the exposure-response relationship between transport noise and IHD. The 'additional risk' represents the added probability of IHD resulting from transport noise exposure above the underlying population probability.

The additional risk of IHD due to transport noise exposure was calculated using the relative risk of IHD due to transport noise exposure and the underlying population probability of IHD.

The underlying population probability of IHD was calculated as per Equation 3 below:

Equation 3 Average probability of IHD

$$Average\ probability\ of\ IHD\ = \frac{population\ IHD\ cases}{total\ population}$$

New Zealand Ministry of Health (2018) estimated the incidence rate per 100,000 for IHD-related mortality and hospitalisations cases. This incidence rate per 100,000 was extrapolated using 2022 population figures to estimate the current rate.

Assumed parameter values relating to the derivation of the average probability of IHD are summarised in Table 5.7 below:

Table 5.7 Average probability of ischaemic heart disease (IHD)

Statistic	Value	Source
New Zealand population (March 2022)	5,127,200	Stats NZ (2022)
IHD mortality (2021 estimate) (Adult population 35+ years only)	227 per 100,000 people	New Zealand Ministry of Health (2018)
IHD hospitalisations (2021 estimate) (Adult population 35+ years only)	1,540 per 100,000 people	New Zealand Ministry of Health (2018)
Average probability of IHD (2021 estimate)	0.702%	

The additional risk of IHD due to noise exposure (per 10 dB increase) was calculated as below:

Equation 4 Additional risk of IHD due to transport noise exposure (per 10 dB)

$$Additional\ risk\ of\ IHD = [(RR\ of\ IHD\ due\ to\ transport\ noise) - 1] * (average\ probability\ of\ IHD)$$

$$Additional\ risk\ of\ IHD_{road\ (central)} = [(1.08) - 1] * 0.702\% = 0.056\%\ (per\ 10\ dB)$$

$$Additional\ risk\ of\ IHD_{rail\ (central)} = [(1.01) - 1] * 0.702\% = 0.007\%\ (per\ 10\ dB)$$

The additional risk of IHD due to noise exposure (per 1 dB increase) was calculated by dividing the additional risk of IHD due to noise exposure by 10 (dB), as calculated in Equation 5 and summarised in Table 5.8:

Equation 5 Additional risk of IHD due to transport noise exposure (per 1 dB)

$$Additional\ risk\ of\ IHD_{road\ (central)} = \frac{0.056\%}{10} = 0.0056\%\ (per\ 1\ dB)$$

$$Additional\ risk\ of\ IHD_{rail\ (central)} = \frac{0.007\%}{10} = 0.0007\%\ (per\ 1\ dB)$$

Table 5.8 Additional risk of ischaemic heart disease (IHD) due to transport noise exposure (per 1 dB)

Health endpoint	Noise source	Noise metric	Additional risk of IHD per 1 dB increase [95% confidence interval]	Range of applicability	Source
IHD Incidence (Adult population 35+ years only)	Road traffic noise	dB L_{den}	Additional risk of IHD = 0.0056% [0.0007% – 0.0105%] x L_{den}	Low = 53 dB	Vienneau et al. (2019)
	Railway noise	dB L_{den}	Additional risk of IHD = 0.0007% [-0.0007% – 0.0021%] x L_{den}	Low = 53 dB	Vienneau et al. (2019)

5.2.4 Health cost valuation assumptions

Health cost parameter values utilised in the cost model were sourced from the literature review. The literature review identified and critiqued existing research relating to health cost valuations, including studies that are specific to New Zealand.

The health cost parameter values derived from the literature review are summarised in this section. Detailed analysis and commentary surrounding the parameter values used in the cost model are provided in the literature review.

Health cost valuations are used to:

- assign DWs to health conditions
- standardise the valuation of mortality and morbidity
- monetise the cost of morbidity and mortality.

5.2.4.1 Disability weights and Quality-Adjusted Life Years (QALYs)

Disability weights (DWs) are used to estimate the severity of a health condition. DWs lie on a scale between zero (indicating the health condition does not impact full health) and one (indicating the health condition results in death). DWs consider both the impact on the length of life and health-related quality of life. DWs support the quantification of Quality-Adjusted Life Year (QALY) by estimating the disutility of a health condition.

The QALY is a measure of health benefits, combining quality of life and life expectancy. QALYs provide a standardised means of valuing the burden of disease. One QALY is the value of a year of life in perfect health.

5.2.4.2 Sleep disturbance

The WHO (2009) established that the DW of sleep disturbance due to environmental noise lies in 0.04–0.1, with a recommended value of 0.07. In effect, this means that being sleep disturbed due to environmental noise reduces a completely healthy individual's health by around 7%. In terms of QALYs, this means that a sleep disturbed individual who is otherwise fully healthy with a life expectancy of 50 years would have 50 x 0.93 = 46.5 QALYs.

In line with the WHO (2009) recommendations, this analysis utilised a DW of 0.07 as its central scenario with sensitivity testing at 0.04 and 0.1.

Table 5.9 Disability weight (DW) for sleep disturbance

DW – sleep disturbance	DW value – disturbance	Source
High	0.1	WHO (2009)
Low	0.04	
Central	0.07	

5.2.4.3 Annoyance

The WHO (2011) established that the DW of annoyance due to environmental noise lies in the range of 0.01–0.12, with a recommended value of 0.02. In effect, this means that being annoyed due to environmental noise reduces a completely healthy individual's health by around 2%. In terms of QALYs, this means that an annoyed individual who is otherwise fully healthy with a life expectancy of 50 years would have 50 x 0.98 = 49 QALYs.

In line with the WHO (2011) recommendations, this analysis utilised a DW of 0.02 as its central scenario with sensitivity testing at 0.01 and 0.12.

Table 5.10 Disability weight (DW) for annoyance

DW – annoyance	DW value – annoyance	Source
High	0.12	WHO (2011)
Low	0.01	
Central	0.02	

5.2.4.4 Ischaemic heart disease

The WHO (2018) recommends that a DW of 0.405 be used for people living with IHD. Living with IHD reduces a completely healthy individual's health by around 40.5%. The DW for death (due to IHD) is 1. In terms of QALYs, an individual living with IHD who is otherwise fully healthy with a life expectancy of 50 years would have 50 x 0.495 = 24.75 QALYs.

In line with the WHO (2018) recommendations, this analysis utilised a DW of 0.405 as its central scenario for people living with IHD.

Table 5.11 Disability weight (DW) for ischaemic heart disease (IHD)

DW – IHD	DW value – IHD	Source
Central	0.405	WHO (2018)
Death	1	

5.2.4.5 Health value monetisation

New Zealand Treasury's CBAx Tool User Guide (2021) includes two different values for quantifying QALYs in monetary terms. Following recommendations from the guide, this analysis considered the Pharmac (2015) derived QALY value as the central scenario and the value of statistical life (VoSL) derived QALY value (New Zealand Treasury, 2021) for sensitivity testing.

Table 5.12 Quality-Adjusted Life Year (QALY) valuations

Parameter	Source value	NZ$ 2022 value	Source
QALY value (central)	$32,258 (2019)	$36,363	Pharmac (2015)
QALY value (high)	$54,213 (2020)	$59,897	New Zealand Treasury (2021)

5.2.4.6 Ischaemic heart disease (IHD) morbidity and mortality assumptions

The average years of life lost due to IHD used in this analysis were calculated as the difference between average life expectancy (New Zealand Ministry of Health, 2022) and the average age at death of people who died of IHD (Crimmins et al., 2008).

The average years of life lived with IHD used in this analysis followed the recommended value outlined in Crimmins et al. 2008.

New Zealand Ministry of Health estimated the incidence rate per 100,000 for IHD-related mortality and hospitalisations cases. This incidence rate per 100,000 was extrapolated using 2021 population figures to estimate the current rate.

Table 5.13 Ischaemic heart disease (IHD) mortality and hospitalisation estimates

IHD statistic	Of total 2021 population	Source
IHD mortality (Adult population 35+ years only)	4,420*	New Zealand Ministry of Health (2018)
IHD hospitalisations (Adult population 35+ years only)	36,001*	New Zealand Ministry of Health (2018)

This indicates an IHD survival rate of 87.72% and a mortality rate of 12.28%. Therefore, QALY calculations in this analysis assumed mortality in 12.28% of modelled IHD cases.

The assumptions pertaining to IHD morbidity and mortality are summarised below in Table 5.14.

Table 5.14 Ischaemic heart disease (IHD) morbidity and mortality assumptions

Parameter	Value	Source
Life expectancy (2017–2019)	82* *midpoint between male life expectancy (80) and female life expectancy (83.5)	Stats NZ (2019)
Age at death (IHD)	72	Crimmins et al. (2008)
Years of life lost due to IHD (2022 estimate)	10	
Years of life lived with IHD	7.3	Crimmins et al. (2008)
IHD survival rate (2021 estimate)	87.72%	New Zealand Ministry of Health (2018)
IHD mortality rate (2021 estimate)	12.28%	New Zealand Ministry of Health (2018)

5.3 Health cost impact methodology

The social impact of transport noise was measured by quantifying the cost of three health endpoints – annoyance, sleep disturbance, and IHD.

Results generated by the noise model were estimated at 1 dB increments and disaggregated by geographic unit for the three health endpoints. Modelled results were used to estimate:

- the number of residents who suffer from one or more of the conditions at each noise interval
- each condition's economic and societal cost as a whole at each noise interval.

The cost model estimates the social cost of noise associated with road and rail across New Zealand by region, district, and health district, and has not been further disaggregated. However, the same cost impact methodology and framework could potentially be applied to estimate the quantitative impact of a change in noise levels in a given area from an infrastructure project.

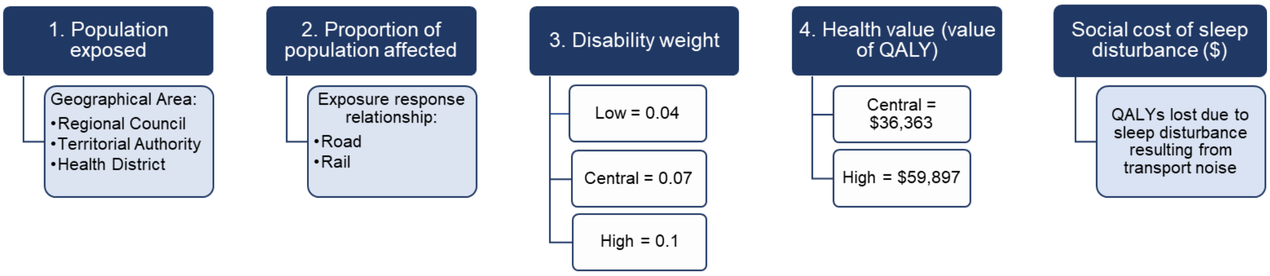
5.3.1 Sleep disturbance

The method used to quantify the social cost of sleep disturbance followed the methodology identified and utilised by DEFRA (2014). The social cost of noise-induced sleep disturbance was quantified as the product of four parts, as described in Equation 6 and illustrated in Figure 5.1:

Equation 6 Social cost of sleep disturbance

$$(1)\ population\ exposed\ *(2)\ proportion\ of\ population\ affected\ *(3)\ DW *(4)\ health\ value$$
$$=\ social\ cost\ of\ sleep\ disturbance\ (\$)$$

Figure 5.1 Social cost of sleep disturbance



The four components of the noise-induced sleep disturbance equation's social cost are explored extensively in previous sections. In summary:

- The *population exposed* parameter measures the number of people exposed to transport noise, disaggregated by geographical area and distinguishable by transport noise type. Population exposed parameter values were captured through noise modelling at 1 dB (L_{night}) increments.
- The *proportion of the population affected* parameter measures the percentage of the population that experiences high levels of sleep disturbance resulting from transport noise (%HSD). %HSD was estimated separately for rail and road transport noise at 1 dB (L_{night}) increments using the exposure-response formulas identified by Basner and McGuire (2018), as recommended in the literature review.
- *DW* is a measure of the severity of health impact which is used to derive QALYs. DWs consider impacts on both longevity and quality of life. As recommended by the WHO (2009), this analysis considered a DW of 0.07 for HSD as its central scenario, with sensitivity testing at 0.04 and 0.1.

- *Health value* refers to the value of one year of life lived in full health (ie, the value of one QALY). This analysis considered the Pharmac (2015) derived QALY value as the central scenario and the value of statistical life (VoSL) derived QALY value (New Zealand Treasury, 2021) for sensitivity testing.

Equation 7 Number of people highly sleep disturbed due to transport noise

$$1 * 2 = Number\ of\ people\ highly\ sleep\ disturbed\ due\ to\ transport\ noise$$

The product of the first two terms was taken to estimate the number of people who experience HSD resulting from transport noise.

The resulting estimate is disaggregated by geographical area, transport noise type, and dB level.

Equation 8 QALYs lost due to HSD

$$(1 * 2) * 3 = QALYs\ lost\ due\ to\ HSD$$

The product of the estimate of the number of people who experience HSD and the DW of sleep disturbance equates to the estimated population QALYs lost due to HSD.

The QALY measures health benefits, combining quality of life and life expectancy. As DWs consider both the impact on the length of life and health-related quality of life, multiplying DWs by the estimated number of people who experience HSD due to transport noise equates to the estimated population QALYs lost due to HSD.

The resulting estimate is disaggregated by geographical area, transport noise type, and dB level.

Equation 9 Value of QALYs lost due to HSD

$$[(1 * 2) * 3] * 4 = value\ of\ QALYs\ lost\ due\ to\ HSD$$

The product of estimated population QALYs lost due to HSD and the recommended monetary value of a QALY was taken to convert the population health cost of transport noise-induced HSD to quantifiable monetary terms. This output was used to represent the health cost of transport noise-induced sleep disturbance.

The resulting estimate is disaggregated by geographical area, transport noise type, and dB level.

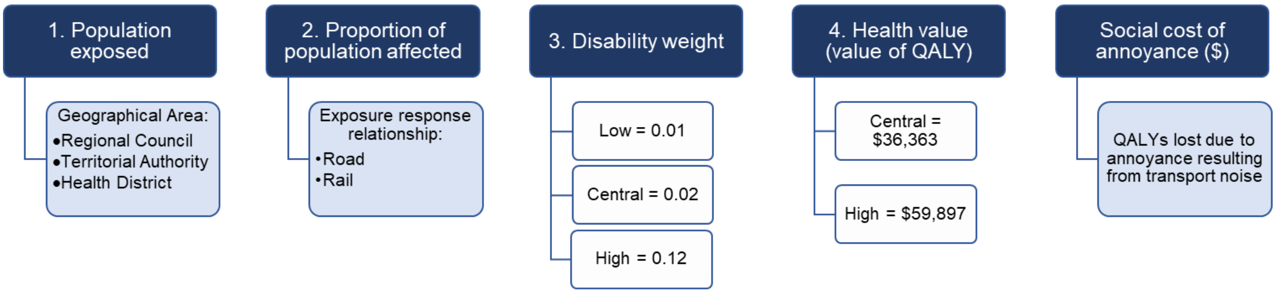
5.3.2 Annoyance

The method used to quantify the social cost of annoyance followed the methodology identified and used by DEFRA (2014). The social cost of noise-induced annoyance was quantified as the product of four parts, as described in Equation 10 and illustrated in Figure 5.2:

Equation 10 Social cost of annoyance

$$(1)\ population\ exposed\ * (2)\ proportion\ of\ population\ affected\ * (3)\ DW * (4)\ health\ value = social\ cost\ of\ annoyance\ (\$)$$

Figure 5.2 Social cost of annoyance



The four components of the noise-induced annoyance equation's social cost are explored extensively in previous sections. In summary:

- The *population exposed parameter* is a measure of the number of people exposed to transport noise, disaggregated by geographical area and distinguished by transport noise type. Population exposed parameter values were captured through noise modelling at 1 dB (L_{den}) increments.
- The *proportion of the population affected* parameter measures the percentage of the population that experiences high levels of annoyance resulting from transport noise (%HA). %HA was estimated separately for rail and road transport noise at 1dB (L_{den}) increments using formulas identified by Guski et al. (2017), as recommended in the literature review.
- *DW* is a measure of the severity of health impact which is used to derive QALYs. DWs consider impacts on both longevity and quality of life. This analysis utilised a DW of 0.02 for HA as its central scenario with sensitivity testing at 0.01 and 0.12, as recommended by the WHO (2011).
- *Health value* refers to the value of one year of life lived in full health (ie, the value of one QALY). This analysis considered the Pharmac (2015) derived QALY value as the central scenario and the value of statistical life (VoSL) derived QALY value (New Zealand Treasury, 2021) for sensitivity testing.

Equation 11 Number of people highly annoyed due to transport noise

$$1 * 2 = Number\ of\ people\ highly\ annoyed\ due\ to\ transport\ noise$$

The product of the first two terms was taken to estimate the number of people who experience HA resulting from transport noise.

The resulting estimate is disaggregated by geographical area, transport noise type, and dB level.

Equation 12 QALY's lost due to HA

$$(1 * 2) * 3 = QALYs\ lost\ due\ to\ HA$$

The product of the estimate of the number of people who experience HA resulting from transport noise and the DW of annoyance equates to an estimation of population QALYs lost due to HA resulting from transport noise.

The resulting estimate is disaggregated by geographical area, transport noise type, and dB level.

Equation 13 Value of QALYs lost due to HA

$$[(1 * 2) * 3] * 4 = value\ of\ QALYs\ lost\ due\ to\ HA$$

The product of estimated population QALYs lost due to HA and the recommended monetary value of a QALY was taken to convert the population health cost of transport noise-induced HA to quantifiable monetary terms. This output was used to represent the health cost of transport noise-induced annoyance.

The resulting estimate is disaggregated by geographical area, transport noise type, and dB level.

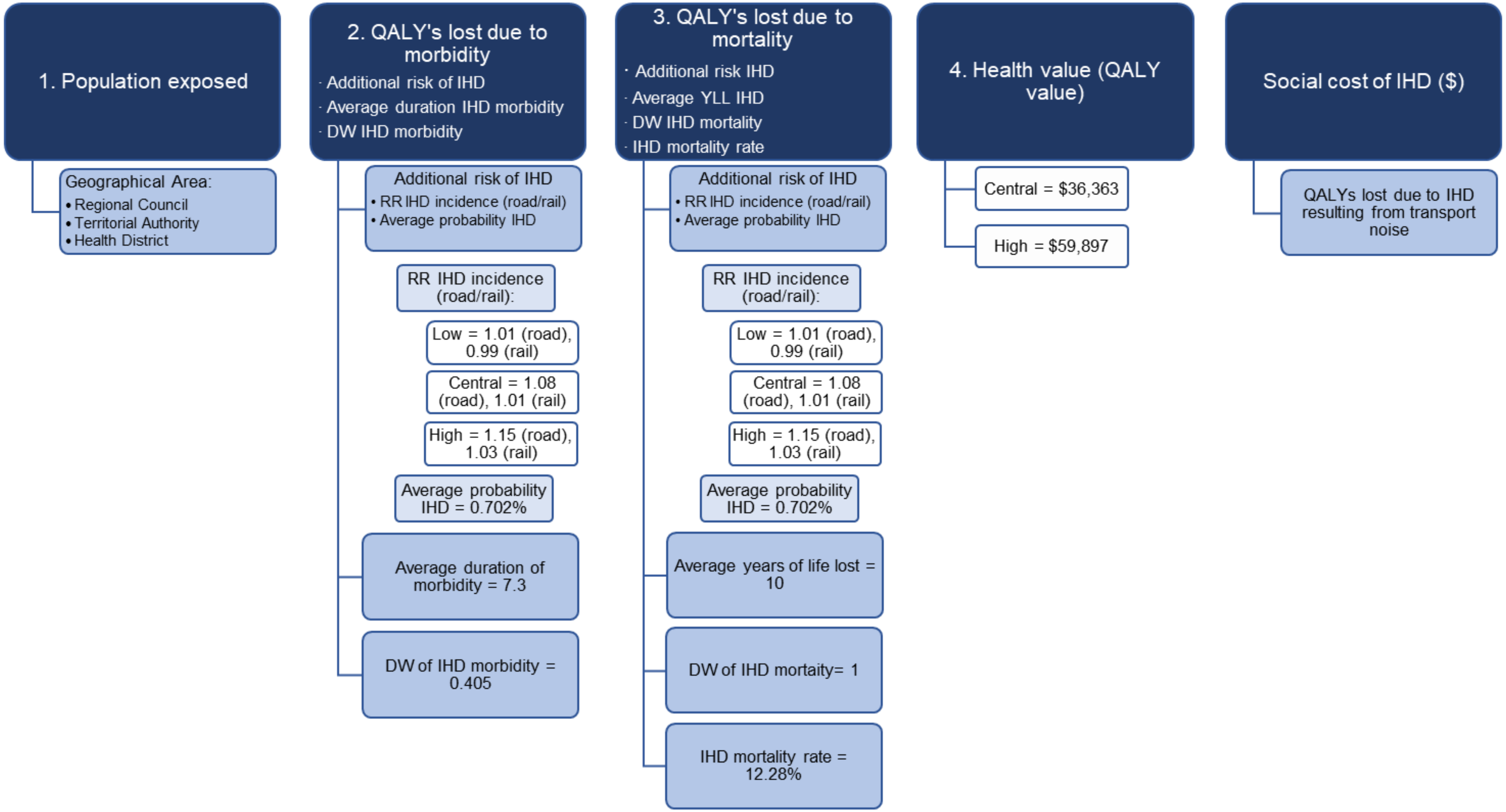
5.3.3 Ischaemic heart disease

The method used to quantify the social cost of IHD followed the methodology identified and utilised by Harding et al. (2011). IHD outcomes are characterised by both mortality and morbidity. As such, the social cost of noise-induced IHD was quantified as described in Equation 14 and illustrated in Figure 5.3:

Equation 14 Social cost of IHD

$$(1)\ population\ exposed * \left((2)\ QALYs\ lost\ due\ to\ morbidity\ + (3)\ QALYs\ lost\ due\ to\ mortality\right) * (4)\ health\ value\ = social\ cost\ (\$)$$

Figure 5.3 Social cost of ischaemic heart disease (IHD)



The components that make up the social value of transport noise-induced IHD are summarised below.

The *population exposed* parameter is a measure of the number of people exposed to transport noise, disaggregated by geographical area and distinguished by transport noise type (road or rail). Population exposed parameter values were captured through noise modelling at 1 dB (L_{den}) increments.

QALYs lost due to morbidity was calculated as described in Equation 15 below.

Equation 15 QALYs lost due to IHD morbidity

$$(2)\ QALYs\ lost\ due\ to\ morbidity = additional\ risk\ of\ IHD\ due\ to\ transport\ noise * average\ duration\ of\ morbidity * DW_{morbidity}$$

The *additional risk of IHD* due to transport noise metric represents the incidence (%) of IHD that can be attributed to transport noise. It considers the population *average probability of IHD* and the *relative risk of IHD* due to transport noise to estimate the added probability of IHD that results from transport noise exposure, above the underlying population probability.

In calculating the additional risk of IHD metric, this analysis derived the average probability of IHD using IHD mortality and hospitalisation data (New Zealand Ministry of Health, 2018) and New Zealand population data (Stats NZ, 2022). As described in Equation 3, the average probability of IHD used in this analysis 0.702%.

As recommended by Vienneau et al. (2019), this analysis considered a relative risk of 1.08 per 10 dB increase for IHD due to road noise as its central scenario, with sensitivity testing at 1.01 and 1.15. For IHD due to rail noise, a relative risk of 1.01 per 10 db increase was used as the central scenario, with sensitivity testing at 0.99 and 1.03.

Equation 4 and Equation 5 detail the calculation of additional risk of IHD due to road and rail noise.

The *average duration of IHD morbidity* statistic used in this analysis was 7.3 years, as recommended by Crimmins et al. (2008).

A *DW* of 0.405 for IHD morbidity was used, as recommended by the WHO (2018).

QALYs lost due to mortality were calculated as follows:

Equation 16 QALYs lost due to IHD mortality

$$(3)\ QALYs\ lost\ due\ to\ mortality = \left(additional\ risk\ of\ IHD\ due\ to\ transport\ noise * average\ years\ of\ life\ lost * DW_{mortality}\right) * IHD\ mortality\ rate$$

The *additional risk of IHD* due to transport noise metric represents the incidence (%) of IHD that can be attributed to transport noise. The derivation of additional risk of IHD due to road and rail noise is explored extensively above and in section 5.2.3 of this report.

The *average years of life lost due to IHD* was calculated as the difference between life expectancy and the average age at death of people who died of IHD (see Table 5.14). The average years of life lost due to IHD used in this analysis was 10 years, as calculated using New Zealand population data (Stats NZ, 2019) and IHD mortality data (Crimmins et al., 2008).

The *DW* of IHD mortality is 1 (death).

The *IHD mortality rate* (%) was calculated as the quotient of total IHD-related deaths on total IHD cases (see Table 5.14). The IHD mortality rate used in this analysis was 12.28%, as calculated using IHD mortality data (New Zealand Ministry of Health, 2018).

Health value refers to the value of one year of life lived in full health (ie, the value of one QALY). This analysis considered the Pharmac (2015) derived QALY value as the central scenario and the value of statistical life (VoSL) derived QALY value (New Zealand Treasury, 2021) for sensitivity testing.

Equation 17 QALYs lost due to IHD

$$1 * (2 + 3) = QALYs\ lost\ due\ to\ IHD$$

The product of population transport noise exposure and QALYs lost due to IHD morbidity and mortality from transport noise exposure equates to population QALYs lost due to IHD resulting from noise exposure. The resulting estimate is disaggregated by geographical area, transport noise type, dB level, morbidity QALYs lost, and mortality QALYs lost.

Equation 18 Value of QALYs lost due to IHD

$$1 * (2 + 3) * 4 = value\ of\ QALY's\ lost\ due\ to\ IHD$$

The product of estimated population QALYs lost due to IHD and the recommended monetary value of a QALY was taken to convert the population health cost of transport noise-induced IHD to quantifiable monetary terms. This output represented the health cost of transport noise-induced IHD.

The resulting estimate is disaggregated by geographical area, transport noise type, dB level, value of morbidity QALYs lost, and mortality QALYs lost.

5.4 Sensitivity analysis

The cost model draws upon a range of assumptions and parameter values that carry varying degrees of uncertainty. To address this uncertainty, provision has been made for sensitivity testing to allow for the assessment of outcomes, in terms of social cost estimates, arising from changes to the model’s underlying assumptions.

Sensitivity analysis was carried out to test the ‘sensitivity’ of outputs to parameter value changes for DWs, relative risk (IHD), and QALY values.

5.4.1 Disability weights

Sleep disturbance: this analysis utilised a DW of 0.07 for sleep disturbance as its central scenario with sensitivity testing at 0.04 (low) and 0.1 (high), as recommended by the WHO (2009).

Annoyance: this analysis utilised a DW of 0.02 for annoyance as its central scenario with sensitivity testing at 0.01 (low) and 0.12 (high), as recommended by the WHO (2011).

Ischaemic heart disease: this analysis utilised a DW of 0.405 for IHD morbidity as its central scenario, as recommended by the WHO (2018). There was insufficient evidence obtained from the literature review to undertake sensitivity testing for the DW of IHD morbidity. The DW for death (due to IHD) is 1.

5.4.2 Relative risk

Road: this analysis considered a relative risk of 1.08 per 10 dB increase for IHD due to road noise as the central scenario, with sensitivity testing at 1.01 and 1.15, as recommended by Vienneau et al. (2019).

Rail: this analysis considered a relative risk of 1.01 per 10 dB increase for IHD due to rail noise as the central scenario, with sensitivity testing at 0.99 and 1.03, as recommended by Vienneau et al. (2019).

5.4.3 QALY values

Following recommendations from the New Zealand Treasury's CBAx Tool User Guide (2021), this analysis considered the Pharmac (2015) derived QALY value of $36,363 as the central scenario and the value of statistical life (VoSL) derived QALY value of $59,897 from the New Zealand Treasury (2021) for sensitivity testing.

5.5 Cost model limitations

In the absence of New Zealand-specific parameter values, the cost model is reliant upon dose-relationship parameter values derived based on evidence derived through studies undertaken globally across Europe, America, and Asia. While it is acknowledged that the derivation and use of New Zealand-specific parameters would be preferable as they would more precisely reflect local conditions, the WHO guidelines suggest that these values can be considered applicable in other regions and are suitable for a global audience (WHO, 2018).

In addition to the uncertainties surrounding the parameter values used in the cost model, the methodology which underpins the model also carries with it several limitations:

- For each health endpoint, the results can only be used to estimate statistical average reactions, not specific responses. For example, sleep disturbance comprises three distinct parts: being unable to fall asleep, being woken up, and being unable to return to sleep. Therefore, depending on the type of sleep disturbance, the valuation may change.
- The methodology adopted in the cost model reduces the potential for overlap between annoyance and sleep disturbance; however, it does not eliminate it.
- Based on the method adopted to quantify the health cost of IHD, the number of cases by geographical area was not able to be determined.

5.6 Cost model results

The tables below provide a summary of the social cost of noise values for road and rail noise at a New Zealand Territorial Authority level in current year prices.

The central scenario below assumes a QALY value of $32,258, and all DWs and relative risk values associated with calculating the social cost of noise for sleep disturbance, sleep annoyance, and ischaemic heart disease set to central values, as described in the sections above.

Note that due to the method adopted to quantify the health cost of IHD, the geographic spread of cases is not known and therefore the number of cases is not able to be presented in the results tables below.

Table 5.15 Social cost of road noise – central case

Social cost of road noise	Sleep disturbance	Sleep annoyance	Ischaemic heart disease
Number of cases	122,711	223,454	N/A
Social cost ($NZ, m) – total	$312 m	$163 m	$27 m
Social cost ($NZ) – per capita	$89.92	$46.78	$7.69

Table 5.16 Social cost of rail noise – central case

Social cost of rail noise	Sleep disturbance	Sleep Annoyance	Ischaemic heart disease
Number of cases	45,737	39,746	N/A
Social cost ($NZ, m) – total	$116 m	$29 m	$7 m
Social cost ($NZ) – per capita	$33.51	$8.32	$2.10

Table 5.17 Social cost of noise – road, Territorial Authority, central case

Territorial Authority	Source of noise – road								
	Number of cases			Social cost ($NZ) – total			Social cost ($NZ) – per capita		
	Sleep disturbance	Sleep annoyance	Ischaemic heart disease	Sleep disturbance	Sleep annoyance	Ischaemic heart disease	Sleep disturbance (adult population)	Sleep annoyance (adult population)	Ischaemic heart disease (35+ years)
Area outside Territorial Authority	0	0	N/A	$-	$-	$-	$-	$-	$-
Ashburton District	646	1252	N/A	$1,644,479	$910,348	$109,022	$66	$37	$6
Auckland	44692	81033	N/A	$113,759,937	$58,931,888	$9,539,119	$99	$51	$12
Buller District	293	494	N/A	$745,562	$359,350	$69,707	$98	$47	$11
Carterton District	167	339	N/A	$425,324	$246,525	$50,511	$61	$35	$9
Central Hawke's Bay District	221	419	N/A	$561,722	$304,589	$55,622	$54	$29	$7
Central Otago District	597	1046	N/A	$1,519,842	$760,769	$106,065	$90	$45	$8
Chatham Islands Territory	0	0	N/A	$-	$-	$-	$-	$-	$-
Christchurch City	11557	22285	N/A	$29,418,109	$16,206,963	$2,506,025	$104	$57	$13
Clutha District	416	767	N/A	$1,059,178	$558,006	$122,859	$80	$42	$12
Dunedin City	3828	6704	N/A	$9,742,860	$4,875,222	$817,699	$103	$52	$12
Far North District	1476	2511	N/A	$3,757,278	$1,825,963	$421,567	$80	$39	$11
Gisborne District	1067	1985	N/A	$2,716,573	$1,443,809	$257,119	$82	$44	$10
Gore District	340	620	N/A	$866,493	$451,221	$79,749	$93	$48	$11
Grey District	358	632	N/A	$911,700	$459,675	$115,295	$91	$46	$15
Hamilton City	4651	8714	N/A	$11,838,042	$6,337,156	$769,777	$103	$55	$10
Hastings District	1295	2448	N/A	$3,295,548	$1,780,268	$301,750	$56	$31	$7
Hauraki District	595	1078	N/A	$1,514,085	$784,239	$159,371	$100	$52	$13
Horowhenua District	927	1747	N/A	$2,360,369	$1,270,422	$208,984	$93	$50	$10
Hurunui District	254	447	N/A	$647,080	$325,320	$52,460	$68	$34	$7
Invercargill City	1591	3005	N/A	$4,050,410	$2,185,253	$389,064	$100	$54	$13
Kaikoura District	134	226	N/A	$339,970	$164,700	$36,871	$109	$53	$15
Kaipara District	517	879	N/A	$1,316,332	$639,497	$162,849	$77	$37	$12
Kapiti Coast District	1321	2354	N/A	$3,361,340	$1,711,655	$246,222	$81	$41	$7
Kawerau District	133	227	N/A	$339,115	$165,243	$25,136	$69	$34	$7
Lower Hutt City	2187	3991	N/A	$5,567,337	$2,902,783	$431,857	$72	$38	$8
Mackenzie District	118	219	N/A	$301,122	$159,339	$14,111	$80	$42	$5
Manawatu District	700	1220	N/A	$1,782,987	$887,195	$172,430	$81	$40	$10
Marlborough District	1135	2091	N/A	$2,887,840	$1,520,618	$265,803	$79	$41	$9
Masterton District	502	958	N/A	$1,277,661	$696,618	$103,826	$67	$37	$7
Matamata–Piako District	729	1362	N/A	$1,854,762	$990,188	$168,234	$73	$39	$9
Napier City	2063	3667	N/A	$5,252,129	$2,666,737	$470,377	$114	$58	$13
Nelson City	1209	2227	N/A	$3,076,830	$1,619,544	$236,718	$79	$42	$8
New Plymouth District	2050	3745	N/A	$5,219,303	$2,723,847	$511,624	$88	$46	$11
Opotiki District	159	275	N/A	$403,716	$200,117	$34,101	$61	$30	$7
Otorohanga District	176	303	N/A	$448,910	$220,235	$36,409	$62	$31	$7
Palmerston North City	2552	4784	N/A	$6,494,917	$3,479,542	$517,136	$106	$57	$13

Source of noise – road									
Territorial Authority	Number of cases			Social cost ($NZ) – total			Social cost ($NZ) – per capita		
	Sleep disturbance	Sleep annoyance	Ischaemic heart disease	Sleep disturbance	Sleep annoyance	Ischaemic heart disease	Sleep disturbance (adult population)	Sleep annoyance (adult population)	Ischaemic heart disease (35+ years)
Porirua City	1438	2562	N/A	$3,661,000	$1,863,570	$323,476	$93	$48	$11
Queenstown Lakes District	1083	1906	N/A	$2,756,828	$1,385,911	$138,500	$89	$45	$7
Rangitikei District	316	561	N/A	$804,915	$408,106	$93,578	$73	$37	$11
Rotorua District	1854	3369	N/A	$4,718,010	$2,449,956	$386,346	$93	$48	$10
Ruapehu District	315	565	N/A	$801,298	$410,886	$43,640	$90	$46	$7
Selwyn District	883	1560	N/A	$2,247,599	$1,134,651	$161,294	$52	$26	$5
South Taranaki District	618	1148	N/A	$1,573,028	$834,719	$164,439	$80	$43	$11
South Waikato District	558	987	N/A	$1,419,557	$718,086	$109,723	$84	$42	$9
South Wairarapa District	228	441	N/A	$579,410	$320,568	$67,399	$71	$39	$10
Southland District	533	927	N/A	$1,357,634	$674,132	$114,535	$60	$30	$7
Stratford District	179	344	N/A	$455,565	$250,360	$40,845	$66	$37	$8
Tararua District	409	758	N/A	$1,041,701	$551,576	$97,843	$80	$42	$9
Tasman District	1219	2180	N/A	$3,102,531	$1,585,744	$335,638	$78	$40	$10
Taupo District	1144	2079	N/A	$2,912,795	$1,511,679	$204,121	$106	$55	$10
Tauranga City	3782	6822	N/A	$9,627,724	$4,961,414	$780,213	$95	$49	$10
Thames–Coromandel District	858	1513	N/A	$2,183,312	$1,100,580	$229,360	$91	$46	$11
Timaru District	1348	2475	N/A	$3,431,221	$1,800,224	$383,542	$97	$51	$14
Upper Hutt City	1108	1978	N/A	$2,819,127	$1,438,238	$201,985	$87	$44	$8
Waikato District	1214	2094	N/A	$3,090,219	$1,522,709	$294,221	$58	$29	$7
Waimakariri District	1134	1973	N/A	$2,885,950	$1,434,845	$275,119	$65	$32	$8
Waimate District	139	241	N/A	$353,981	$174,970	$40,212	$59	$29	$8
Waipa District	1224	2234	N/A	$3,115,642	$1,624,434	$275,016	$80	$42	$9
Wairoa District	160	280	N/A	$406,255	$203,943	$34,221	$70	$35	$8
Waitaki District	513	946	N/A	$1,304,932	$688,305	$125,269	$77	$41	$9
Waitomo District	202	365	N/A	$513,475	$265,449	$47,477	$78	$40	$9
Wellington City	4557	8276	N/A	$11,599,531	$6,018,507	$891,576	$75	$39	$9
Western Bay of Plenty District	637	1108	N/A	$1,621,339	$806,084	$161,430	$42	$21	$5
Westland District	220	375	N/A	$559,904	$272,712	$54,067	$82	$40	$10
Whakatane District	768	1339	N/A	$1,953,647	$973,723	$178,225	$77	$38	$9
Whanganui District	983	1803	N/A	$2,501,216	$1,311,033	$263,931	$74	$39	$10
Whangarei District	2431	4191	N/A	$6,187,268	$3,047,867	$645,961	$93	$46	$13
Total annual cost per health endpoint				$312,345,476	$162,509,825	$26,728,671			

Table 5.18 Social cost of noise – rail, Territorial Authority, central case

Source of noise – rail									
Territorial Authority	Number of cases			Social cost ($NZ) – total			Social cost ($NZ) – per capita		
	Sleep disturbance	Sleep annoyance	Ischaemic heart disease	Sleep disturbance	Sleep annoyance	Ischaemic heart disease	Sleep disturbance (Adult population)	Sleep annoyance (Adult population)	Ischaemic heart disease (35+ years)
Area outside Territorial Authority	0	0	N/A	$-	$-	$-	$-	$-	$-
Ashburton District	492	372	N/A	$1,252,735	$270,724	$87,414	$51	$11	$5
Auckland	11559	10159	N/A	$29,421,097	$7,388,029	$1,791,102	$25	$6	$2
Buller District	209	198	N/A	$531,786	$143,657	$40,444	$70	$19	$6
Carterton District	192	198	N/A	$487,908	$144,095	$44,731	$70	$21	$8
Central Hawke's Bay District	278	224	N/A	$708,335	$162,676	$48,512	$68	$16	$6
Central Otago District	0	0	N/A	$-	$-	$-	$-	$-	$-
Chatham Islands Territory	0	0	N/A	$-	$-	$-	$-	$-	$-
Christchurch City	2716	2208	N/A	$6,914,515	$1,605,977	$457,123	$25	$6	$2
Clutha District	370	295	N/A	$942,269	$214,411	$59,634	$71	$16	$6
Dunedin City	2009	1674	N/A	$5,114,205	$1,217,709	$285,482	$54	$13	$4
Far North District	0	9	N/A	$-	$6,727	$3,216	$-	$0	$0
Gisborne District	0	0	N/A	$-	$-	$-	$-	$-	$-
Gore District	390	309	N/A	$991,676	$224,475	$64,919	$106	$24	$9
Grey District	221	202	N/A	$562,757	$146,813	$34,305	$56	$15	$4
Hamilton City	1297	1001	N/A	$3,301,973	$728,261	$167,372	$29	$6	$2
Hastings District	446	334	N/A	$1,135,136	$242,839	$77,289	$19	$4	$2
Hauraki District	0	0	N/A	$-	$-	$-	$-	$-	$-
Horowhenua District	945	865	N/A	$2,404,356	$629,403	$151,075	$95	$25	$7
Hurunui District	209	171	N/A	$532,970	$124,483	$32,985	$56	$13	$4
Invercargill City	659	433	N/A	$1,676,324	$314,786	$120,024	$42	$8	$4
Kaikoura District	208	192	N/A	$528,569	$139,647	$32,706	$169	$45	$13
Kaipara District	0	13	N/A	$-	$9,458	$3,237	$-	$1	$0
Kapiti Coast District	1174	1034	N/A	$2,987,834	$751,981	$203,255	$72	$18	$6
Kawerau District	0	0	N/A	$-	$-	$-	$-	$-	$-
Lower Hutt City	2022	2165	N/A	$5,147,804	$1,574,655	$387,577	$67	$21	$7
Mackenzie District	0	0	N/A	$-	$-	$-	$-	$-	$-
Manawatu District	1051	984	N/A	$2,676,402	$715,291	$144,825	$122	$32	$8
Marlborough District	587	489	N/A	$1,495,101	$355,382	$106,129	$41	$10	$4
Masterton District	183	186	N/A	$464,641	$135,505	$41,342	$24	$7	$3
Matamata–Piako District	563	442	N/A	$1,432,907	$321,760	$90,971	$57	$13	$5
Napier City	863	757	N/A	$2,196,070	$550,437	$153,972	$48	$12	$4
Nelson City	0	0	N/A	$-	$-	$-	$-	$-	$-
New Plymouth District	140	101	N/A	$355,994	$73,093	$27,259	$6	$1	$1
Opotiki District	0	0	N/A	$-	$-	$-	$-	$-	$-
Otorohanga District	169	122	N/A	$430,718	$88,917	$23,479	$60	$12	$5
Palmerston North City	1404	1163	N/A	$3,574,191	$845,621	$204,424	$58	$14	$5

Source of noise – rail									
Territorial Authority	**Number of cases**			**Social cost ($NZ) – total**			**Social cost ($NZ) – per capita**		
	Sleep disturbance	**Sleep annoyance**	**Ischaemic heart disease**	**Sleep disturbance**	**Sleep annoyance**	**Ischaemic heart disease**	**Sleep disturbance (Adult population)**	**Sleep annoyance (Adult population)**	**Ischaemic heart disease (35+ years)**
Porirua City	1029	874	N/A	$2,619,892	$635,497	$127,729	$67	$16	$5
Queenstown Lakes District	0	0	N/A	$-	$-	$-	$-	$-	$-
Rangitikei District	702	644	N/A	$1,785,699	$468,512	$94,511	$162	$43	$11
Rotorua District	0	0	N/A	$-	$-	$-	$-	$-	$-
Ruapehu District	858	726	N/A	$2,183,882	$527,631	$109,254	$246	$59	$16
Selwyn District	804	674	N/A	$2,045,362	$489,911	$118,039	$47	$11	$4
South Taranaki District	291	154	N/A	$739,520	$112,039	$43,448	$38	$6	$3
South Waikato District	521	396	N/A	$1,326,197	$288,285	$85,896	$78	$17	$7
South Wairarapa District	127	141	N/A	$322,666	$102,822	$31,292	$39	$13	$5
Southland District	221	199	N/A	$561,620	$144,397	$40,719	$25	$6	$2
Stratford District	165	94	N/A	$419,588	$68,360	$23,996	$61	$10	$5
Tararua District	407	318	N/A	$1,037,211	$231,556	$68,640	$80	$18	$7
Tasman District	0	0	N/A	$-	$-	$-	$-	$-	$-
Taupo District	0	0	N/A	$-	$-	$-	$-	$-	$-
Tauranga City	1979	1749	N/A	$5,038,039	$1,271,715	$303,349	$50	$13	$4
Thames–Coromandel District	0	0	N/A	$-	$-	$-	$-	$-	$-
Timaru District	496	432	N/A	$1,262,669	$314,069	$93,780	$36	$9	$3
Upper Hutt City	1002	1083	N/A	$2,550,690	$787,833	$199,158	$78	$24	$8
Waikato District	1889	1599	N/A	$4,807,213	$1,162,654	$242,433	$91	$22	$6
Waimakariri District	589	464	N/A	$1,498,254	$337,770	$107,314	$34	$8	$3
Waimate District	94	83	N/A	$239,195	$60,459	$15,006	$40	$10	$3
Waipa District	290	211	N/A	$737,609	$153,481	$46,685	$19	$4	$2
Wairoa District	0	0	N/A	$-	$-	$-	$-	$-	$-
Waitaki District	571	474	N/A	$1,453,221	$344,565	$105,038	$86	$20	$8
Waitomo District	283	206	N/A	$721,013	$149,788	$40,527	$109	$23	$8
Wellington City	2087	1701	N/A	$5,311,545	$1,236,896	$281,641	$34	$8	$3
Western Bay of Plenty District	719	689	N/A	$1,830,003	$501,153	$105,522	$47	$13	$3
Westland District	0	0	N/A	$-	$-	$-	$-	$-	$-
Whakatane District	257	248	N/A	$654,285	$180,561	$31,557	$26	$7	$2
Whanganui District	0	0	N/A	$-	$-	$-	$-	$-	$-
Whangarei District	0	287	N/A	$-	$208,738	$87,092	$-	$3	$2
Total annual cost per health endpoint				$116,413,646	$28,905,504	$7,287,429			

5.6.1 Sensitivity analysis results

The cost model draws upon a range of assumptions and parameter values that carry varying degrees of uncertainty. To address this, sensitivity analysis was carried out to test the 'sensitivity' of outputs to parameter value changes for DWs, relative risk (IHD), and QALY values.

Aside from the central case scenario (presented above), the following additional scenarios were assessed:

- **Low scenario** – utilising 'low' DW assumptions for sleep disturbance and annoyance, and 'low' relative risk factor for IHD.
- **VoSL QALY value** – which utilises a value of statistical life (VoSL) derived QALY value of $59,897 from the New Zealand Treasury (2021), compared to the Pharmac (2015) derived QALY value of $36,363 used in the central scenario.
- **High scenario** – utilising 'high' DW assumptions for sleep disturbance and annoyance, 'high' relative risk factor for IHD and high QALY (VoSL derived) value.

The results of this sensitivity analysis are provided in Table 5.159 and Table 5.16 below.

Table 5.19 Social cost of road noise sensitivity analysis

Social cost of road noise ($NZ, m)	Sleep disturbance	Sleep annoyance	Ischaemic heart disease	Total cost per scenario
Central case	$312 m	$163 m	$27 m	$502
Low scenario	$178 m	$81 m	$3 m	$262
VoSL QALY value	$514 m	$268 m	$44 m	$826
High scenario	$735 m	$1,606 m	$83 m	$2,424

Table 5.20 Social cost of rail noise sensitivity analysis

Social cost of rail noise ($NZ, m)	Sleep disturbance	Sleep annoyance	Ischaemic heart disease	Total cost per scenario
Central case	$116 m	$29 m	$7 m	$152
Low scenario	$67 m	$14 m	$–	$81
VoSL QALY value	$191 m	$48 m	$12 m	$251
High scenario	$274 m	$286 m	$36 m	$596

5.6.2 Uncertainty in cost modelling

As shown in the sensitivity analysis results above, the low scenario estimate of cost from road traffic noise deviates from the central scenario by -48%, and the high scenario estimate of cost deviates from the central scenario by +383%. For rail noise, the low scenario estimate of cost from road traffic noise deviates from the central scenario by -47%, and the high scenario estimate of cost deviates from the central scenario by +292%.

Therefore, it is considered that the sensitivity analysis reflects the greatest amount of uncertainty in the study as a whole, that is, the uncertainty inherent in the selection of disability weightings and QALY values.

6 Conclusion

Continued exposure to land transport noise can be associated with a wide range of adverse impacts on human health.

The purpose of this project was to estimate and monetise these impacts on the New Zealand economy. This has been done through identification of literature relevant to this study, recommendation of metrics to use as a result of the health review, modelling of road traffic and rail noise across New Zealand, and cost modelling using the outcomes of the noise modelling and health review.

The results of the analysis estimate that exposure to road noise costs the New Zealand economy approximately $502 million a year, as a result of its impact on the health outcomes of the community, while exposure to rail noise costs the New Zealand economy a further $153 million a year.

This is likely a conservative estimate of the true cost of road and rail noise on the community given that a number of additional costs were not estimated as part of this study due to insufficient evidence and data to support the quantification of those impacts.

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7.1 Noise model references

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Appendix A: Literature review

The literature review for the SCON project was undertaken in three parts to focus on the noise modelling, health implications, and social costs of land transport noise exposure in New Zealand. Each is described in its own subsection.

A.1 Noise model literature review

The information presented covers the approach to acoustic modelling and how this is relevant to SCON. This includes the calculation methods, indicators, and tools that are available to calculate and present the noise levels that could be appropriate to determine the impact of transport noise in New Zealand.

A.1.1 Approach

The literature review was an important step of the initial research stage for SCON as it allowed the specialists developing the transport noise exposure model to:

- identity existing scientific research and regulatory approaches to transportation noise modelling
- determine the applicability of findings to New Zealand conditions
- confirm that the modelling methods proposed within the SCON project scope and methodology are best for the project
- highlight where there are gaps in current understanding and where further work could be undertaken.

The information considered includes:

- the different noise indicators used to describe transportation noise impacts
- calculation methodologies used locally and abroad for calculating road and rail noise levels
- indicators used to support the quantification of human health and economic impacts from transportation noise in New Zealand
- methods typically used to convert the noise modelling indicators to ensure consistency across different transportation types
- the status of strategic noise mapping and good practice guidelines from the EU.

The various literature reviewed in this report has been sourced from the NZTA, New Zealand Standards, scientific journal papers, UK government departments – DEFRA, European Economic Area and Australian Councils; all of which are referenced at the end of this document.

A.1.2 Clarifications

The calculation of road traffic noise levels is based on modelling guidance developed by the NZTA. The applicability of alternative calculation methods was not considered in this review.

The scope of the project does not include the prediction of airport or seaport noise levels, so methods used to calculate transportation noise from airports or seaports have not been considered in this review. A further literature review of these transportation noise sources may be undertaken if information that can be used for this study becomes available.

A.1.3 Noise Indicators

Background

Different indicators are often used to interpret noise depending on the type, duration, and the receptor. The study of human response to noise has resulted in the development of noise descriptors in terms of frequency and time weightings. These describe and replicate human response to noise and its impact (enHealth, 2018).

For example, an acoustical average ($L_{eq,T}$) or statistical value (ie, $L_{10,T}$) may be suitable for describing the contribution from a steady, continuous noise source over time whereas a short impulsive event may be better described as a discrete, maximum value (L_{max}).

Similarly, the long-term exposure to environmental noise may be best described using indicators that reflect the combined measured or modelled contribution from different noise sources for the time periods under investigation (L_{den} and L_{night}). These noise exposure indicators have been effectively used for strategic noise mapping studies that aim to establish a relationship between noise exposure, health, and economic impacts. Table A.1 lists common indicators used to interpret sources of environmental noise.

Table A.1 Existing use of noise indicators

Noise Indicator	Description
$L_{A10,18h}$	A-weighted centile level equalled or exceeded for 10% of an 18-hour time interval. This is the noise descriptor used for interpreting road traffic noise in accordance with the Calculation of Road Traffic Noise (CRTN) prediction method.
$L_{Aeq,18h}$	A-weighted equivalent continuous sound pressure level during an 18-hour period expressed in dB. This is the basic noise descriptor used for rail noise in the Calculation of Rail Noise (CRN) prediction method.
$L_{Aeq,24h}$	A-weighted equivalent continuous sound pressure level during a 24-hour period expressed in dB. The preferred noise descriptor used for road traffic noise assessment in New Zealand in accordance with New Zealand Standard 6806:2010 *Acoustics – Road Traffic Noise – New and Altered Roads* (NZS 6806). A correction of −3 dB is applied to $L_{A10,18h}$ for conversion to $L_{Aeq,24h}$.
L_{den}	A-weighted over a 24-hour period that includes a penalty of the evening and night-time level to account for the elevated noise sensitivity in these time periods (as defined in section 3.6.4 of ISO 1996-1:2016).
L_{day}	Equivalent continuous sound pressure level when the reference time interval is the day (07:00–19:00). A component to calculate L_{den}. Also known as $L_{Aeq,12h}$
$L_{evening}$	Equivalent continuous sound pressure level when the reference time interval is the evening (19:00–23:00). A component to calculate L_{den}. Also known as $L_{Aeq,4h}$
L_{night}	Equivalent continuous sound pressure level when the reference time interval is the night (23:00–07:00). A component to calculate L_{den}. L_{night} is widely used for exposure assessment in health effect studies.
$L_{Aeq,t}$	A-weighted equivalent continuous sound pressure level during a stated time interval (t), expressed in dB.

Challenges can arise when the selected environmental noise modelling methods produce results using different indicators to describe the impact from different noise sources. When this happens, indicators calculated for one or more of the sources would typically be converted using an empirical rule or formula to derive a common indicator that can be used for the study.

For example, in New Zealand, traffic noise is described using the $L_{Aeq,24h}$ indicator. A sound pressure level described using this indicator would not be immediately compatible with the indicators used to assess the long-term exposure to environmental noise such as the L_{den} and would require an adjustment as described in the following section.

Road traffic noise

The 'Environmental Noise Guidelines for the European Region' provides guidelines for converting indicators used to describe traffic noise ($L_{A10,18h}$ and $L_{Aeq,24h}$) and noise exposure indicators (L_{den} and L_{night}). The empirical conversion factors were largely developed by the UK's Transport Research Laboratories (TRL) using field studies that were targeted at identifying the correlation between different measured noise indicators (TRL, 2006[16]).

TRL's work identified three methods for conversion between road traffic and noise exposure indicators. This includes the consideration of the different noise level corrections that would apply to motorway or non-motorway roads (to account for the different noise profile produced by these types of roads) for UK conditions.

Naish et al. (2011) identified the correlation with the $L_{A10,,18h}$ indicator in relation to Australian conditions, and investigated the conversion methods to L_{den} based on his field measurements study, specific to Queensland, for comparison.

Brink et al. (2018) also identified a general set of rules to convert road traffic noise derived from AADT based on studies from within Europe with the associated uncertainties. A comparison of the applied conversion methods is shown in Table A.2.

Table A.2 Noise level indicator adaption comparison

Modelled indicator [$L_{A10(18h)}$]	Desired indicator							
	$L_{Aeq,24h}$ SPLAN	$L_{Aeq,24h}$ Naish	L_{Night} Naish	L_{den} Naish	L_{day} DefraTRL[17]	$L_{Evening}$ Defra TRL	L_{Night} Defra TRL Method 3	L_{den} Defra TRL Method 3
Predicted level	−3 dB	−1 dB	−6 dB	+3 dB	−1 dB	0 dB	−9 dB	0 dB

A conversion method between modelled outputs and noise exposure indicators has not been developed specifically for New Zealand road conditions. However, the information reviewed suggests that the DEFRA TRL Adaption Method 3 (conversion between modelled $L_{Aeq,\ 24h}$ and the L_{den}) has not been superseded by a process that would be more suitable for similar desktop assessments.

Furthermore, this conversion was recently adopted for the *National Land Transport (Road) Noise Map* (AECOM, 2019). It is considered appropriate for use in this method for SCON to maintain consistency with the European approach and recent, similar mapping undertaken in New Zealand.

Rail noise

The conversion of modelled train noise levels to noise exposure indicators is recommended. This is because of the many variables from line to line that can influence the noise being produced over a typical 24-hour period of operation. Even if track properties remain constant, the frequency of services, the speed of operation, and the composition of the train fleet (eg, electric, diesel commuter, and freight) make it difficult to apply a global correction that could apply across the network.

[16] L_{den} indicator is A-weighted over a 24-hour period split between day, evening, and night periods. With a 10 dB penalty applied to the night and a 5 dB penalty added to the evening. The penalties are to account for people's increased sensitivity during the evening and night.

[17] For DEFRA by TRL and Casella – Method for converting the UK road traffic noise index $L_{A10(18h)}$ to the EU noise indices for road noise mapping, 2006.

The preferred method is to calculate levels for L_{day}, $L_{evening}$ and L_{night} using the relevant breakdown of train movements, on each rail line, for each time interval, prior to calculating the L_{den}. Train movements in this format have been used for SCON.

Gaps and further work

The key gaps within the current literature is an endorsed method for converting modelled noise levels to noise exposure indicators for New Zealand conditions.

Further work could consider the development of an accepted method for calculating noise exposure indicators in New Zealand. An extensive field study would be valuable in quantifying known corrections that can be used for future projects.

A.1.4 Calculation methodologies

Road traffic noise

The NZTA refer to New Zealand Standard 6806:2010 *Acoustics – Road Traffic Noise – New and Altered Roads* (NZS 6806) for the assessment of traffic noise.

This standard includes the following items relevant to this study:

- The calculation method to be used to assess traffic noise impacts as sensitive receptors (UK Department of Transport and Welsh Office, 'Calculation of Road Traffic Noise', 1988 (CRTN)).
- Accepted adjustments of the CRTN method to account for the noise level indicators preferred in New Zealand (converting $L_{10,18h}$ to $L_{Aeq,24h}$).
- Road surface corrections to account for the noise profiles associated with different asphalt types.
- Standardised acoustic terminology.

The NZTA has also released several guidance documents to support the implementation of NZS 6806. This includes the 'Guide to state highway noise mapping'[18] and 'Guide to assessing road-traffic noise using NZS 6806'[19]. They prescribe the technical methods that can be adopted by acoustic consultants undertaking a road traffic noise assessment in accordance with NZS 6806.

It is considered appropriate for road traffic noise predictions to be carried out in general accordance with the above documents given that methods for traffic noise modelling and noise mapping have been established and refined by the NZTA to align with the practices adopted in other parts of the world.

Rail noise

The assessment of noise impacts caused by rail noise are routinely carried out within Europe, North America, parts of Asia, and Australia. The method for predicting operation varies from region to region. For instance, in Australia, Kilde 67/130 (developed in Norway and superseded by NORD 2000) is the more commonly used method. The UK prefers to use the UK Department of Transport Calculation of Railway Noise (CRN) method and Germany uses the Schall 03 method. All these calculation methods have been developed or modified to allow for noise predictions to be carried out using 3D noise modelling software.

Several studies and research papers have gone into detail about these methods, which countries they are used in, and the key calculation differences. HJA Leeuwen (1999) compared different railway noise prediction models within Europe, looking at the source types, source heights, and source locations. Leeuwen

[18] NZTA Draft Guide to state highway noise mapping, November 2013, Version 1.0

[19] NZTA, Guide to assessing road-traffic noise using NZS 6806 for state highway asset improvement projects, August 2016, Version 1.1

found that whilst the core calculation of rail noise is the same for all methods, the differences occur with the corrections applied and the source term set up.

To improve consistency and comparability of methods used to assess environmental noise exposures, the EU developed a standard method for noise mapping, named Common Noise Assessment Methods in Europe (CNOSSOS-EU). CNOSSOS-EU is an extensive methodology guidance which defines exactly the roads, rail, airport transport, and industrial noise that should be included within noise maps. These include the type of noise sources, location of source, reference conditions, and an extensive list of prescriptive methods for calculating sound power levels and corrections to be applied. However, the CNOSSOS-EU recognises that other input parameters are required but are only significant in specific local situations. Whilst this method was specifically developed to fulfil Member States' obligations under Environmental Noise Directive (END), it is understood that adaption to this method has been difficult due to its data intensive nature.

All methods account for the ground condition and the reflection of sound from a known surface or structure. However, the assumptions and complexity of calculations for each method do vary and can produce different results for the same modelling conditions.

Consequently, the choice of calculation method is dependent on the intended purpose and requirement. This idea is demonstrated further by the Australian state authorities' preference of Kilde 67/130, even though this method has been superseded with the more recent and complex version of the Nordic method NORD 2000.

CRN was selected for SCON and has been incorporated into the scope and methodology of the study. The decision was based on the benefits that a less complex modelling method would provide (eg, faster calculation speeds, more readily available modelling inputs, and easier data handling) compared to the more complex methods available.

Although train types and operations differ, the origins of the rail network within New Zealand is largely based on that built in the UK. Rail gauge (spacing of the rails on a railway track) are generally different, however the ballast, sleeper, and fixing mechanisms are similar. Therefore, the measurement of New Zealand rolling stock forms an important stage of the research project, allowing noise produced by New Zealand rolling stock to be accurately captured in the noise model.

Calculation of Rail Noise (CRN)

This calculation method was first developed in the UK in 1993 and is still widely used today. CRN was developed as a tool to model new and additional railways to identify properties that were entitled to noise abatement.

The modelling inputs that CRN uses to calculate the level of noise produced by each modelled train pass-by includes:

- the sound power level produced by each type of rolling stock
- the maximum speed assumed for each train type on each individual line
- track corrections to account for the rail construction, including the presence and type of bridges
- the distance of track radius throughout the network to determine penalties applied to account for curve squeal.

The train line is modelled as a line source that incorporates the above items where relevant.

The reduction of noise over distance from the train line noise source to the receiver location considers the:

- atmospheric attenuation
- ground attenuation
- screening attenuation

- loss or increase of noise caused by reflection or absorption.

Although the CRN method has been considered appropriate for SCON, there are known limitations (Hardy, 2004) that include the inability to accurately calculate noise from:

- high-speed trains
- idling trains
- more than three possible noise source heights – rolling noise at the head of the nearest rail and at two metres or four metres above the rail head
- single maximum noise events produced during a train pass-by or when warning horns operate
- scenarios that include multiple reflections (sound waves ‘bouncing’ off surfaces) from complex geometry or surfaces.

The above items are largely irrelevant to the methodology adopted for calculating rail noise as part of SCON strategic mapping.

Applicability to New Zealand

Government regulators or agencies responsible for managing the current New Zealand rail network do not have a preferred approach to rail noise assessment in New Zealand. While work has been carried out for new rail expansions using the CRN method (Wiri to Quay Park third main rail line – 2020)[20], rail noise has also been managed by way of ‘reverse sensitivity’ for new building developments within 40 metres of existing rail corridors.

For this reason, the methodology for SCON was developed by adopting a fit-for-purpose approach that has been proven in other parts of the world and would be suitable for informing the health cost model (as described in section 3).

Although the development and use of a more complex and integrated model has been used in Europe (CNOSSOS-EU), Member States were allowed to use their own national methods up until 2019 for the development of strategic noise maps as part of the END (2015). This signals the time and effort that has been required to promote, develop, and integrate a common approach to noise modelling over the past 10 years.

At this stage, the same level of work has not been undertaken in the southern hemisphere to achieve this degree of integration. Accordingly, while much of the CNOSSOS-EU procedure is very helpful for developing a methodology for strategic noise mapping outside of Europe, other countries, such as New Zealand, require more work to be undertaken in this area before adopting the same approach.

Gaps and further work

One of the key gaps noted throughout this part of the review was the lack of an endorsed rail noise assessment guideline. Government regulators and/or agencies throughout the world often advise on the acceptable approach to measuring and modelling operational rail noise.

Further considerations for New Zealand conditions could include:

- the development of a preferred calculation methodology and accepted train noise corrections for assessments
- applicable noise criteria to protect the health and wellbeing of the community
- an external acoustic criterion at sensitive receivers

[20] Marshall Day Acoustics, Wiri to Quay third main rail line, noise and vibration assessment, RP 001 20200311, 10 July 2020

- a wider monitoring exercise of the New Zealand rolling stock
- a supporting noise level database for train types across the New Zealand network that can be used for predictive modelling.

A longer-term goal could be to investigate the use of a common noise assessment method similar to that currently used for strategic noise modelling in Europe.

A.1.5 Strategic noise mapping

The first set of strategic noise maps commissioned in New Zealand were for the Auckland motorway network (220 kilometres of state highway) in 2009. This followed recommendations published in the Land Transport New Zealand Research Report 299 that described noise mapping as one of the options available to help address the problems associated with transport noise.

The NZTA initiated further work in 2012 to update the noise mapping undertaken in this area of the network to address the shortcomings of the initial assessment[21]. The key outcome of this being the identification of areas where traffic noise mitigation would be beneficial to the community.

The conclusions and lessons learnt from these earlier studies led to the development and publication of the New Zealand 'Guide to state highway noise mapping' (2013). The breadth of the strategic noise mapping was expanded further in 2019 when AECOM was commissioned to model and map the traffic noise produced by the national road network.

Whilst this type of area-wide strategic noise mapping has not been legislated in New Zealand, it has now been a requirement of the Member States of the European Union for nearly 20 years. As such, the approach to strategic noise mapping adopted throughout the EU is generally more advanced than other parts of the world.

The END provides the EU's Member States with a common approach for managing the potentially adverse effects of environmental noise on their populations. It requires the production of noise maps to determine noise exposure in the community that is caused by major modes of transportation and industrial sources. These maps serve as the basis of local action plans and for measures to reduce noise emissions.

To pursue its stated aims, the END focuses on three action areas that include:

- determining the communities' exposure to environmental noise
- ensuring that information on environmental noise and its effects is made available to the public
- preventing and reducing environmental noise where necessary and preserving the acoustic environment where existing exposure is acceptable.

The END assessment methods describe the indicators to be used for strategic noise mapping in Article 7 and Annex IV. A concise summary of how they are used has been provided below:

- The L_{den} and L_{night} noise indicators are to be produced to determine health impact relationships.
- Dose-effect relations should be assessed using the L_{den} noise indicator.
- The relations between sleep disturbance and noise should be assessed using the L_{night} indicator.
- Separate strategic noise maps to be made for road traffic, rail traffic, aircraft, and industrial noise for the agglomerations.
- Strategic noise maps may be presented as graphical plots, numerical data in tables, or in electronic form.

[21] Hannaby. R, Chiles. S, Worts. C, Whitlock. J, Haigh. A, State Highway Noise Mapping – Auckland Motorways Case Study - 26th ARRB conference – Research driving efficiency, Sydney, NSW 2014

Not all of this information is relevant to SCON, however key documents with information that would be most useful was identified and includes:

- methodological guidance for estimating burden of disease from environmental noise
- Good Practice Guide for Strategic Noise Mapping and the Production of Associated Data on Noise Exposure, 2006
- Environmental Noise in Europe, 2020
- Common Noise Assessment Methods in Europe (CNOSSOS-EU)
- NZTA Guide to state highway noise mapping.

From these documents, solutions and practices applicable to SCON and CRN method have been incorporated into the methodology.

Gaps and further work

The key gap within the current literature is an endorsed method for strategic noise mapping of railways or industry in New Zealand.

Further work could consider the development of an accepted method for railway and industry noise mapping in New Zealand, including the derivation of cumulative impacts, similar to the NZTA guide for road traffic.

A.1.6 Conclusions

A literature review was undertaken to investigate the approach to transportation noise modelling in other regions throughout the world. The focus was on the different calculation methods, indicators, and tools that are available to calculate and present data that could be appropriate to determine the impact of transport noise in New Zealand.

It wasn't unexpected to find that the European Union, encouraged by their legislated END requirements, continue to be a leader in the field of strategic noise mapping. Accordingly, the approach to strategic noise mapping adopted throughout the EU is generally more advanced than New Zealand and other parts of the world.

Technical publications detailing the lessons learnt by Member States over the past 20 years of the END have provided the project with a useful set of resources. This has helped to improve on the methods previously used for noise mapping projects commissioned by the NZTA between 2009 and 2019. These items have been incorporated into the acoustic methodology for a more systematic approach.

The acoustic indicators used to describe noise exposure impacts have not changed since the most recent strategic noise mapping project undertaken by the NZTA in 2019. The latest publications from the WHO continue to refer to the L_{den} and L_{night} indicators to describe health impacts from long-term exposure to environmental noise and sleep disturbance respectively.

The use of these indicators means that a conversion of the modelled output from the acoustic calculation software would need to be undertaken for the traffic noise predictions. NZTA's NZS 6806 noise assessment standard has been established and refined to align with the practices adopted in other parts of the world but was not found to provide a specific conversion factor or value. In this case the DEFRA TRL Adaption Method 3 has been considered the most appropriate method for converting modelled traffic noise levels to noise exposure indicators based on the information available.

A rail noise assessment method has not been published in New Zealand at the time of this research. However, a review of the different prediction methods indicates that results suitable for strategic noise mapping can be achieved using the methods developed in the UK or Europe.

CRN was selected based on the benefits that a less complex modelling method would provide (eg. faster calculation speeds, more readily available modelling inputs, and easier data handling) compared to the more complex methods available. It would also allow for noise levels to be calculated for L_{den} and L_{night} for the purpose of this project.

The main gap found in the current literature was an accepted method for calculating noise from transportation other than roads in New Zealand. Future investigations into the development of standards or guidelines could provide useful support for projects of a similar nature.

A.2 Health review literature review

A.2.1 Purpose

In relation to the assessment of health effects from transport noise, a literature review has been undertaken to identify existing relevant research which has been completed in the area, including studies that are specifically relevant to New Zealand.

The literature review has been undertaken to identify the studies that require critical review and evaluation in the health review (further described in section 3). As such the literature review presented here has not provided the outcomes of the critical review of the literature identified.

A.2.2 Approach

The literature review has been undertaken in two stages.

Stage 1: Existing understanding based on critical review

The first part of the literature review has been to identify key robust existing critical reviews that represent the current understanding and robust exposure-response relationships for the evaluation of health impacts from transport noise.

Most specifically, the key recent critical review of studies and the evidence in relation to health effects of noise, including transport noise, is from the WHO (2018). The WHO (2018) review provides a summary of the outcomes of detailed systematic reviews of large number of studies (Heroux & Verbeek, 2018), completed using the GRADE system, and published in peer-reviewed journals. In general, these reviews are considered to be robust and have considered a large number of studies published in the period 2000–2015. It is noted that there is some discussion in the literature in relation to the robustness of the WHO review on noise annoyance (Gjestland, 2019a, 2019b, 2020; Guski et al., 2017; Guski et al., 2019). These reviews are considered to be a key source of data and quantitative exposure-response relationships relevant to characterising the health effects of transport noise.

In addition to the systematic review completed by the WHO (2018), enHealth (2018) has also undertaken a systematic review of the literature in relation to the health effects of noise. The enHealth (2018) review also used the GRADE system to review the studies and determine the strength of evidence in relation to health effects and the quantitative exposure-response relationships available.

Stage 2: Additional literature

The WHO (2018) and enHealth (2018) reviews are the most current published systematic reviews (using the GRADE system) of studies relating to the health effects of noise. These included a large number of key studies.

This review has not repeated the systematic review of studies evaluated by the WHO (2018) or enHealth (2018). Hence the literature review has focused on the identification of published studies and reports that supplement the WHO (2018) and enHealth (2018) evaluations.

In addition, it is not considered appropriate to include studies that were published prior to 2000, unless the study is considered to be a key study that has not been updated or further reviewed since 2000. Consideration of studies prior to 2000 may be relevant for the assessment of noise annoyance, based on current commentary in the literature (Gjestland 2019a, 2019b, 2020; Guski et al., 2017; Guski et al., 2019), however where these studies are adopted in regulatory environments they have not been further critically reviewed in this assessment. The reason for the inclusion of a time scale is to ensure the noise studies are likely to reflect current noise sources and exposures, as the characteristics of transport sources (engines, construction, aerodynamics etc) may have changed over time.

This commentary and the selection of the most appropriate exposure-response relationships for characterising health impacts will be considered in the detailed review of this literature.

The focus of Stage 2 of the literature review has therefore been on the following:

- Studies that have been published on the health effects of noise, post 2015.
- Key studies that have been published in the period 2000–2015 (and prior to 2000 for noise annoyance) that were not included in the WHO (2018) or enHealth (2018) reviews. These are identified through review of agency reviews and evaluations of noise.
- Studies where there is sufficient information provided to enable review, and outcomes relate to noise sources without co-exposure with other significant sources such as air quality.
- Studies that are specific to the New Zealand environment.

The literature review has been conducted using online search engines, utilising search terms applicable for the identification of studies on the health effects of transport noise.

A.2.3 Existing understanding based on critical review

The starting point for the literature review are the critical reviews completed by the WHO (2018) and enHealth (2018), which have incorporated studies published to 2015. These are key reviews as they have undertaken systematic reviews of studies using the GRADE system to establish strength and quality of evidence in relation to health effects from noise.

The enHealth (2018) conducted a systematic review of literature in relation to noise and sleep disturbance, cardiovascular disease, and cognition. The review is an update of a previous evaluation (enHealth, 2004) that considered more than 200 papers from the period 1994 to 2014. This review concluded the following:

- There is sufficient evidence of a causal relationship between environmental noise and sleep disturbance and cardiovascular disease.
- **Sleep disturbance**: based on the studies in relation to the effects of noise on sleep disturbance, a night-time evidence-based threshold of 55 dBA at the façade as L_{night} is suggested. There is likely an exposure-response relationship for the effects of noise in sleep, with some studies indicating these relationships begin after 42 dBA L_{max}. The physiological changes reported at these levels should not be considered a threshold for adverse health effects.
- **Cardiovascular effects**: based on studies in relation to the effects of noise on cardiovascular effects, a day-time evidence-based limit of 60 dBA outside as L_{day} is suggested. Variability in study design, quality, adjustments for confounders, and reporting of outcomes makes establishing exposure-response relationships difficult.

- There is some evidence of an association with cognitive performance, including reading comprehension, memory, and attention, however the findings are mixed. There is insufficient evidence of a causal effect of environmental noise on persistent cognitive or learning deficits.
- It is plausible that specific modes of transport (aircraft, road, rail) have different effects on sleep and cardiovascular health, however the evidence is not conclusive.
- It is possible that health impacts are greater in certain vulnerable groups, however further evaluation is required.
- More research is required in specific jurisdictions, which would include Australia and New Zealand.
- There is a lack of research on impacts in rural areas.

The WHO (2018) review conducted a systematic evaluation of literature relating to the following aspects in relation to environmental noise (Heroux & Verbeek, 2018):

- effects on sleep
- annoyance
- cognitive impairment, mental health and wellbeing
- cardiovascular disease, diabetes, and metabolic diseases
- hearing impairment and tinnitus
- adverse birth outcomes
- effectiveness of interventions to reduce exposure to, or adverse outcomes from, environmental noise.

In relation to transport noise, the WHO (2018) identified specific health effects where the evidence was considered sufficiently strong (ie, causal or strongly associated) in relation to exposure to transport noise. For these effects, exposure-response relationships have been developed and recommended by the WHO (2018) for the quantification of effects. Table A.3 presents a summary of the health effects for which exposure-response relationships have been characterised by the WHO (2018).

Table A.3 Summary of key outcomes from WHO (2018) critical review for transport noise

Health outcome	Noise metric	Lowest level of exposure (dB)	Exposure-response relationship per 10 dB increase (RR = relative risk or OR = odds ratio) [95% confidence interval]	Quality of evidence
Road traffic noise				
Cardiovascular effects				
Incidence of ischaemic heart disease (IHD)	L_{den}	53	RR = 1.08 [1.01-1.15]	High
Incidence of hypertension	L_{den}	N/A	RR = 0.97 [0.90-1.05]	Low
Annoyance (as % highly annoyed, %HA)	L_{den}	40	OR = 3.03 [2.59-3.55]	Moderate
Cognitive impairment (as reading and oral comprehension)	L_{den}	N/A	Not established	Very low
Sleep disturbance (as % highly sleep disturbed, %HSD)	L_{night}	43	OR = 2.13 [1.82-2.48]	Moderate
Railway noise				
Cardiovascular effects – incidence of hypertension	L_{den}	N/A	RR = 0.96 [0.88–1.04]	Low
Annoyance (%HA)	L_{den}	34	OR = 3.53 [2.83–4.39]	Moderate
Sleep disturbance (%HSD)	L_{night}	33	OR = 3.06 [2.38–3.93]	Moderate

Health outcome	Noise metric	Lowest level of exposure (dB)	Exposure-response relationship per 10 dB increase (RR = relative risk or OR = odds ratio) [95% confidence interval]	Quality of evidence
Aircraft noise				
Cardiovascular effects				
Incidence of ischaemic heart disease (IHD)	L_{den}	47	RR = 1.09 [1.04–1.15]	Very low
Incidence of hypertension	L_{den}	N/A	RR = 1.00 [0.77–1.30]	Low
Annoyance (as % highly annoyed, %HA)	L_{den}	33	OR = 4.78 [2.28–10.05]	Moderate
Cognitive impairment (as reading and oral comprehension)	L_{den}	55	1 to 2-month delay per 5 dB increase	Moderate
Sleep disturbance (as % highly sleep disturbed, %HSD)	L_{night}	35	OR = 1.94 [1.61–2.33]	Moderate

The exposure-response relationships outlined above utilise noise metrics or measures of L_{night} and L_{den}. These are the most common noise metrics utilised and considered in health studies, along with L_{day}.

A.2.4 Literature search strategy

A considerable body of literature is available that addresses the impact of noise on human health and amenity, with applied studies now being regularly published in health, environmental, and economics journals. An equally large body of literature exists either in the form of working papers and guidelines or in ‘grey’ form as doctoral and master’s dissertations. Both published and grey sources have been considered in this review.

Online databases were searched to identify additional literature and studies using the following search terms:

- papers published from 2016 to 2021 (March 2021); and
- include the term noise; and
- include the following terms (mix of terms used to focus review of studies relating to the health effects of transport noise):
    - environmental*, transport, road, rail, shipping, aircraft, noise, disturbance*, nuisance*, exposure*
    - annoyance
    - health*, burden of disease, cardiovascular disease, acute myocardial infarction (AMI), sleep disturbance, hypertension, stroke, dementia, cognition
    - dose-response, dose-effect, exposure-response
    - model*, methodology*
    - review*, study*, meta-analysis, longitudinal.
- papers specific to the New Zealand population.

* Note that some search terms were very general and required use with a number of other key terms to focus the search onto papers that related to studies suitable for assessing and quantifying health effects of transport noise in the community.

The databases searched were as follows (in order of use):

- Web of Science

- Pubmed
- Google Scholar (noting that this database identified many thousands of references, hence more careful refinement of the above search terms was required)
- ScienceDirect
- Scopus
- Ingentia
- JSTOR

Other databases checked for additional papers include:

- Interscience
- Index to Theses
- REPEC
- SSRN
- EconLit
- PapersFirst
- ProceedingsFirst

A.2.5 Refining the search results

Once literature was identified with the search terms outlined above, studies specifically relevant to characterising health effects of transport noise, and in some cases environmental noise (in general) were selected.

The focus of the search relates to epidemiological studies; hence ecosystem, animal, or cell studies were excluded. A number of papers were eliminated as they were not relevant to this topic (with papers relating to occupational noise exposures, co-exposures of noise and air pollution, the modelling of noise exposure (and not characterising health effects) excluded).

To ensure that the literature review identifies studies that are relevant to the consideration of noise impacts from transport, an initial review and ranking of the studies has been undertaken to identify the following selection criteria:

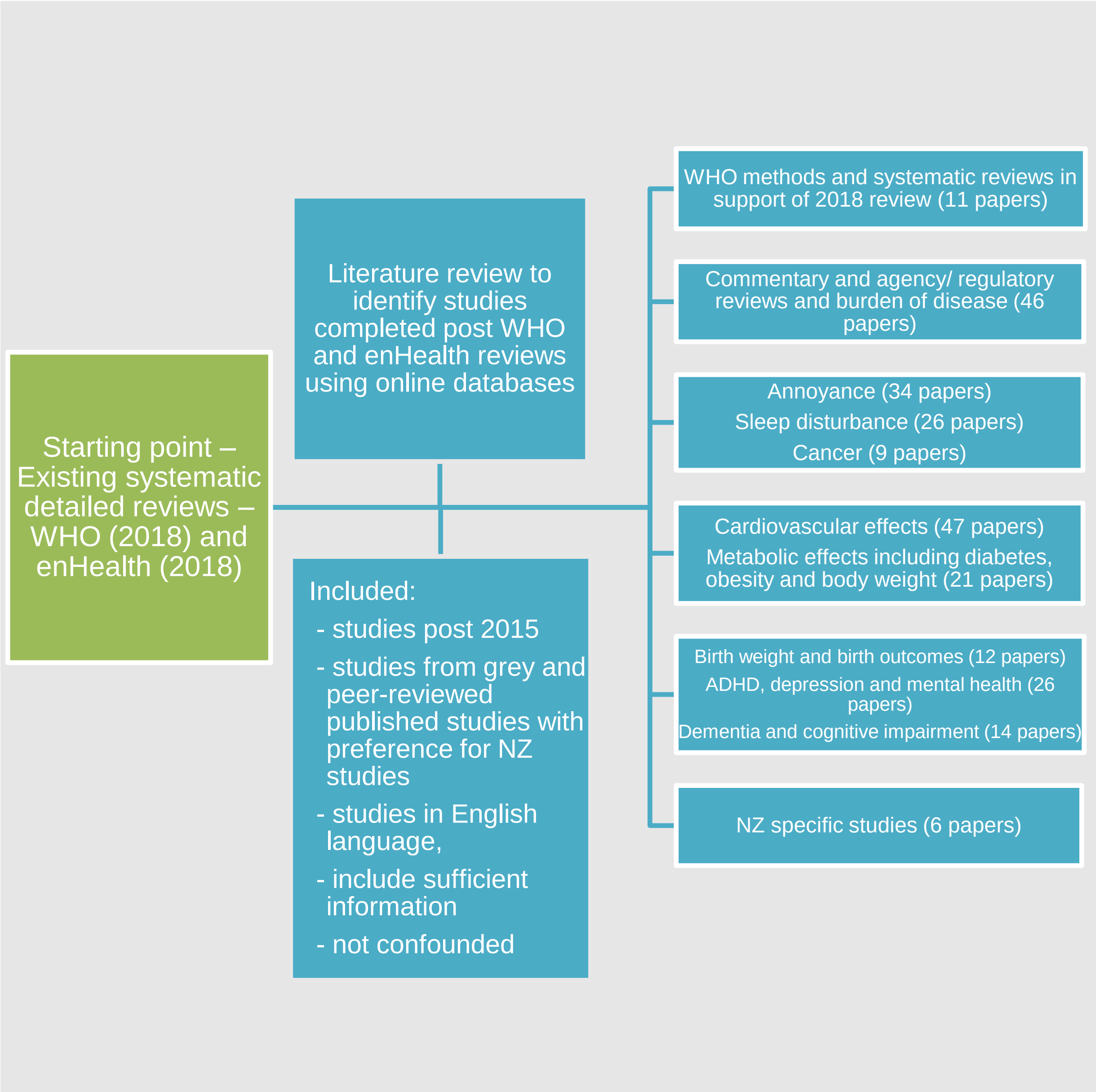
- **Key papers and publication date** – given the potentially large volume of available studies relating to noise and health outcomes, the key studies identified in robust critical reviews already available that relate to health effects of transport noise are the first priority for inclusion. In addition, the review will focus on other more recent studies (eg, from 2000 onwards).
- **Source** – preference has been given to studies published in peer-reviewed journals. However, it is also recognised that very recent studies may be of very high quality but have not yet been through peer review for publication. These have been included where they provide additional information, over and above that obtained.
- **Geography and language** – the review has drawn on the international literature but only those studies relevant to, or transferable to, the New Zealand context will be considered. The review is limited to studies published in English.
- **Sufficient information** – the review or study has sufficient information in relation to the noise source, exposure, population, and health measures.

A.2.6 Literature identified

Figure A.1 presents an overview of the approach adopted for the identification of literature for inclusion in the health assessment, and the number of papers identified in this process. It is noted that some of the papers relate to more than one health outcome and hence have been included multiple times as relevant to the categories listed below. The papers identified are further listed below Figure A.1 with the detailed references included in the reference list relevant to the health review.

These papers and studies have been further considered and critically evaluated in the health review. The final number of papers that require critical review using the GRADE system is a smaller subset of all these papers.

Figure A.1 Overview and outcome of literature review to identify health studies



Papers identified in literature review:

WHO systematic reviews of studies

(Basner & McGuire, 2018; Brown & van Kamp, 2017; Clark et al., 2018a, 2018b; Eriksson et al., 2018; Guski at al., 2017; Heroux & Verbeek, 2018; Nieuwenhuijsen et al., 2017; Śliwińska-Kowalska & Zaborowski, 2017; van Kempen et al., 2018; WHO, 2018)

The WHO evaluation is an update of former reviews (WHO, 1999, 2009, 2011).

Commentary and agency/regulatory reviews

(Argacha et al. 2019; Basner et al. 2017; Basner, Riggs & Conklin 2020; Bistrup et al. 2001; Brink 2020; Bruno et al. 2017; Daiber et al. 2019; DEFRA 2014; Dick 2020; EEA 2014, 2020; enHealth 2004, 2018; ENNAH 2013; Gille, Marquis-Favre & Morel 2016; Gjestland 2019a, 2019b, 2020; Gjestland 2021; Grollman, Maerin & Mhonda 2020; Gupta et al. 2018; Guski et al. 2019; Hahad et al. 2019; Hoffmann 2018; ICCAN 2020; Khan et al. 2018; Kopsch 2016; Münzel, T. et al. 2018a; Münzel, T. & Sørensen 2017; Münzel, T. et al. 2017; Münzel, T. et al. 2018b; Peris & Fenech 2020; Peters et al. 2018; Recio, A. et al. 2016; Roberts & Neitzel 2019; Schäffer et al. 2017; Singh, Kumari & Sharma 2018; Sørensen, Mette et al. 2020; Sparrow et al. 2020; Stansfeld, S. et al. 2021; Tao et al. 2020; Torjesen 2017; UK CAA 2020; Wothge, J. & Niemann 2020; Zeynab et al. 2018)

Annoyance

(Arsalan, Parvin & Abbas Rahimi 2016; Badihian et al. 2020; Bartels, Rooney & Müller 2018; Baudin et al. 2020; Baudin et al. 2018b, 2021; Berry & Flindell 2009; Bouzid, Derbel & Elleuch 2020; Brink et al. 2019a; Carugno et al. 2018; Cerletti et al. 2020; Di et al. 2019; Dzhambov, A. M. et al. 2019; Eze, I. C. et al. 2018; Gille & Marquis-Favre 2016; Grelat et al. 2016; Hong et al. 2018; Jensen, Rasmussen & Ekholm 2018; Lechner, C., Kirisits & Bose-O'Reill 2020; Lechner, Christoph, Schnaiter & Bose-O'Reilly 2019; Lefevre et al. 2020; Nguyen et al. 2018; Paiva, Cardoso & Zannin 2019; Park et al. 2018; Ragettli et al. 2015; Riedel et al. 2019; Schreckenberg et al. 2016; Sieber et al. 2018; Stansfeld, Stephen et al. 2021; Sung et al. 2017; Sung et al. 2016; Wolf et al. 2020; Wothge, Jördis et al. 2017; Zhang & Ma 2021)

Sleep disturbance

(Bartels et al. 2021; Basner, Witte & McGuire 2019; Berry & Flindell 2009; Bodin et al. 2015; Brink et al. 2019b; Carugno et al. 2018; Di et al. 2019; Douglas & Murphy 2016; Elmenhorst, E-M et al. 2019; Evandt et al. 2017; Holt et al. 2015; Joost et al. 2018; Kim, K et al. 2019; Kwak et al. 2016; Martens et al. 2018; Müller, Uwe et al. 2016; Müller, U. et al. 2017; Nassur et al. 2019a; Nassur et al. 2019b; Paiva, Cardoso & Zannin 2019; Park et al. 2018; Perron et al. 2016; Pultznerova et al. 2018; Schreckenberg et al. 2016; Skrzypek et al. 2017; Trieu et al. 2019)

Cardiovascular effects

(Andersson, EM et al. 2020; Ascari et al. 2017; Badihian et al. 2020; Bai et al. 2020; Basner, Riggs & Conklin 2020; Begou, Kassomenos & Kelessis 2020; Bodin et al. 2016; Cai, Y et al. 2018; Carugno et al. 2018; Chen et al. 2021; Dimakopoulou et al. 2017; Dzhambov, A. M. & Dimitrova 2018; Enoksson Wallas, A et al. 2019; Enoksson Wallas, AK 2019; Foraster, Maria et al. 2017; Fu et al. 2017; Fuks et al. 2016; Halonen et al. 2017; Hanigan, I et al. 2019; Hanigan, IC et al. 2019; Héritier et al. 2018; Hoffmann et al. 2015; Kawada 2021; Lechner, C., Kirisits & Bose-O'Reill 2020; Lechner, Christoph, Schnaiter & Bose-O'Reilly 2019; Monrad et al. 2016; Nassur et al. 2019b; Pyko et al. 2019; Recio, A. et al. 2016; Recio, Alberto et al. 2017; Saucy et al. 2020; Seidler, Andreas et al. 2016b, 2016a; Seidler, Andreas et al. 2016c; Seidler, AL et al. 2018; Shin et al. 2020; Stansfeld, SA & Shipley 2015; Thacher et al. 2020; Torjesen 2017; Weihofen et al. 2019; Zeeb et al. 2017; Zijlema, W et al. 2016; Zur Nieden et al. 2016b, 2016a)

Diabetes and body weight

(Basner, Riggs & Conklin 2020; Clark, Charlotte et al. 2017; Dimakopoulou et al. 2017; Eze, Ikenna C et al. 2017; Jørgensen et al. 2019; Kim, A et al. 2017; Pedersen, Marie et al. 2017; Roswall, Nina et al. 2018; Shin et al. 2020; Thacher et al. 2021; Vienneau, Danielle et al. 2019; Zare Sakhvidi, MJ et al. 2018)

(An et al. 2018; Cai, YT et al. 2020; Christensen et al. 2016a; Christensen et al. 2016b; Cramer et al. 2019; Foraster et al. 2018; Oftedal et al. 2015; Pyko et al. 2017; Weyde et al. 2018)

Birth weight and birth outcomes

(An et al. 2018; Argys, Averett & Yang 2020; Christensen et al. 2016a; Christensen et al. 2016b; Dzhambov, A. M. & Lercher 2019a; Enoksson Wallas, AK 2019; Hjortebjerg et al. 2018; Min, K-B & Min, J-Y 2017; Pedersen, M. et al. 2017; Poulsen et al. 2018; Smith et al. 2017; Weyde et al. 2018)

Depression and mental health and Attention Deficit Hyperactivity Disorder (ADHD)

(Badihian et al. 2020; Baudin et al. 2018a; Beutel, Manfred E et al. 2020; Beutel, M. E. et al. 2016; Díaz et al. 2020; Dreger et al. 2015; Dzhambov, A. M. & Lercher 2019b; Dzhambov, A. M. et al. 2019; Eze, Ikenna C. et al. 2020; Generaal et al. 2019; Hammersen, Niemann & Hoebel 2016; Hegewald, J. et al. 2020; Jensen, Rasmussen & Ekholm 2018; Klompmaker et al. 2019; Lan et al. 2020; Leijssen et al. 2019; Okokon et al. 2018; Orban et al. 2016; Schreckenberg et al. 2017; Seidler, A. et al. 2017; Skrzypek et al. 2017; Stansfeld, Stephen et al. 2021; Wright et al. 2018; Zare Sakhvidi, F et al. 2018a, 2018b; Zijlema, WL et al. 2020)

Dementia and cognitive impairment

(Andersson, J et al. 2018; Carey et al. 2018; Carmona et al. 2018; Clark, C., Crumpler & Notley 2020; Culqui et al. 2017; Díaz et al. 2018; Huang et al. 2021; Linares et al. 2017; Robinson et al. 2021; Spilski et al. 2017; Tzivian et al. 2017; Tzivian et al. 2020; Weuve et al. 2020; Yu et al. 2020)

Cancer

(Andersen et al. 2018; Hansen 2017; Hegewald, Janice et al. 2017; Roswall, N. et al. 2018; Roswall, N. et al. 2016; Roswall, N. et al. 2017; Roswall, Nina et al. 2015; Sørensen, M. et al. 2015; Sørensen, Mette et al. 2021)

Specific to New Zealand

(Briggs, Mason & Borman 2015; Dravitzki, Walton & Wood 2006; Humpheson & Wareing 2019; Kjellstrom & Hill 2002; Shepherd, D. et al. 2020; Shepherd, Daniel et al. 2013)

A.3 Cost model literature review

A.3.1 Cost model literature review protocol

Purpose

The purpose of this review protocol is to describe all of the decisions regarding how the literature review for the cost model was undertaken – from the retrieval of literature to the synthesis of review findings. A systematic review methodology was applied to help ensure that as much of the available, relevant research (utilising international and UK studies) evidence as possible was retrieved.

Key research questions to be addressed

The principal aim of the SCON project is to estimate and visualise transport noise exposure in New Zealand and to provide a tool for estimating and visualising the related social (health, hedonic, productivity, and cognitive) cost.

The specific objectives are to:

- collate and understand models/tools/methods currently available to estimate the impact of transport noise in New Zealand
- develop a methodology to estimate the impact of transport noise in New Zealand
- develop a transport noise exposure model (geospatial maps)

- determine health, productivity, cognitive, and hedonic costs (together, ‘social costs’) of transport noise exposure
- develop an integrated tool/model, including geospatial representation, combining noise exposure with the related social costs
- prepare a manual outlining the steps required to update the integrated tool/model.

Ultimately, the outputs of this project are intended to enhance knowledge and understanding of the social costs of transport noise pollution in New Zealand by providing the information and a tool to support appraisal and evaluation in support of policy- and decision-making.

The aim of the social cost model is to provide a means for estimating (quantifying and valuing) the direct and indirect economic effects on society (social costs) associated with prolonged exposure to transport noise. It will be underpinned by exposure-response relationships that link noise levels (by source) to specific health outcomes and will be integrated into a spatially explicit transport noise exposure model.

In this light, a series of key research questions were formulated to guide the literature review process. These were as follows:

- What are the components of social cost associated with prolonged exposure to transport noise?
- What models, tools, and methodologies have been developed and used elsewhere for quantifying and valuing the burden of disease (eg, premature mortality and morbidity) arising from transport noise, what are their relative strengths and weaknesses, and how applicable are they to the New Zealand context?
- What approaches, methodologies, and data sources have been used for valuing the direct and indirect costs of illness (eg, hospitalisation, treatment costs, household assistance, etc)?
- What approaches, methodologies, and data sources have been used for measuring and valuing productivity losses? And to what extent can these be combined with approaches to valuing the burden of disease (eg, using value of a statistical life) without the risk of double-counting?
- What approaches, methodologies, and tools have been applied for valuing the impacts of transport noise exposure on amenity? And to what extent (if any) might these approaches or the values they generate result in double-counting?

The Review Protocol

Search strategy

The search strategy covers the resources to be searched and the search terms to be employed. The aim of the search strategy is to identify as many relevant sources of information as possible in order to ensure that the research questions can be fully addressed. However, a careful balance needs to be struck between the sensitivity of the search strategy (which should identify as much relevant material as possible), and the need to focus the search in order to both exclude irrelevant material and to contain the amount of information retrieved.

Resources searched

A considerable body of literature addresses the impact of noise on human health, amenity, and productivity, with applied studies now being regularly published in health, environmental, and economics journals. An equally large body of literature exists either in the form of working papers and guidelines or in ‘grey’ form as doctoral and master’s dissertations. Both published and grey sources were accessed.

Apart from published academic literature, materials produced by Centres of Excellence, Research Institutes and other relevant organisations within New Zealand and internationally were also accessed. A particular focus was given to approaches and methodologies that have practical application and that have been used to support policy- and decision-making elsewhere. Key sources of information in this regard were considered likely to include the WHO, the Organisation for Economic Co-operation and Development (OECD), the European Commission, the UK's Interdepartmental Groups on Costs and Benefits (Noise Subject Group), Public Health England, the UK Department for Transport's Transport Appraisal Guidance (TAG), the UK Independent Commission on Civil Aviation Noise (ICCAN), the UK Civil Aviation Authority (CAA), Australian Transport Assessment and Planning (ATAP) guidance, and Te Tai Ōhanga – The Treasury.

Consideration was also given to studies that focus on economic valuation of loss of life (morbidity) and changes in quality of life (mortality) more generally (ie, not specifically related to transportation noise exposure) as the approaches and methodologies that have been applied are considered to be equally relevant and transferable to the context of transportation noise.

The search relied extensively on use of the internet to both identify and obtain relevant studies.

Search terms

Each search engine and database was searched using a range of keywords. Keywords, used either singularly or in combination, included:

- Environmental / Transport / Road / Rail / Shipping / Aircraft / Noise / Disturbance / Nuisance / Exposure
- Amenity / Annoyance / Health-related quality of life
- Health / Burden of disease / cardiovascular disease / Acute myocardial infarction / sleep disturbance / hypertension / stroke / dementia / cognitive impairment / cognition
- Dose-response / Dose-effect / Exposure-response
- Mortality / morbidity / quality of life / years lost life / years living with disability / Disability-Adjusted Life Year (DALY) / quality adjusted life year (QALY)
- Value Statistical Life (VoSL) / Value Life Year (VoLY) / Cost of illness
- Model / Methodology / Tool
- Economic / Social / Cost / Impact / Valuation

Study selection criteria and procedures

Study selection criteria were developed to assist with providing a review that is coherent and manageable. The aim of the study selection criteria was to identify those articles that could help to answer the review questions.

The study selection procedure consisted of several stages. Initially, the criteria were applied to the citations generated from searching. This helped identify those studies believed to be relevant and for which full copies of the papers, articles, or book chapters should be obtained. Once copies had been obtained and the abstracts or summaries reviewed, the inclusion/exclusion criteria were again applied, and decisions were made about inclusion of each study in the detailed review.

The selection criteria are described below:

- Geographical coverage and language: the review drew upon the international literature but only those studies relevant to, or transferable to the New Zealand context were considered. The review was limited to studies published in English.
- Publication date: the literature search included studies published from 2000 onwards although appropriate studies cited in the target literature before this year were also explored where these had not been updated and where they are still widely regarded as good or best practice. This takes into consideration the significant advances made over the past two decades in techniques for valuing the social costs of health and other impacts.
- Source and quality: preference was given to studies and approaches that have been through a rigorous review process including papers published in academic journals, technical reports produced or

commissioned by government departments, and other relevant documents that are identified or recognised as official or best practice guidance.

Inventory of retrieved studies

Each of the studies accessed was recorded in an Excel database. The database includes meta-data about the study (eg, authors, date, geographical focus, health and other impacts considered, valuation basis), and any other information considered relevant. The database also contains active hyperlinks to the original study where these are available.

Synthesis of extracted evidence

The aim of the synthesis is to collate and summarise the raw findings of the review (ie, the information stored in the database) into a format that is both useful and relevant to the subsequent development of the social cost model. The findings of the literature review are synthesised in the section of the report on the cost model literature review. The key themes that are covered are the:

- components of the social cost of noise
- approaches to quantifying and valuing health impacts
- approaches to quantifying and valuing productivity losses
- approaches to valuing amenity within social cost of noise assessments.

The literature review report and accompanying database may be considered significant outputs of the study in their own right and can be owned and updated by the NZTA as required.

A.3.2 Background

Continued exposure to transport noise can be associated with a wide range of adverse impacts on human health, quality of life, wellbeing, public amenity, productivity, and ecosystems. Although the strength of evidence varies by noise source, there is at least some evidence linking exposure to persistent or high levels of transport noise to annoyance, sleep disturbance, cardiovascular diseases, metabolic effects, and cognitive functioning. To allow governments to make informed decisions and implement policies for the management and control of noise, suitable appraisal and evaluation tools must be developed which allow both the scale of the problem and the impacts of such policies to be assessed.

Work on quantifying the adverse effects of noise on people has progressed rapidly in the past decade, as highlighted in the most recent WHO Environmental Noise Guidelines (WHO, 2018). The Guidelines are underpinned by a series of peer-reviewed systematic reviews of the pertinent literature since the publication of the WHO Night Noise Guidelines for Europe in 2009 (WHO, 2009) and a more recent review by van Kamp et al. (2020). There are also increasingly sophisticated ways of quantifying these impacts in terms of their effects on morbidity and mortality (WHO, 2011; Brown & Van Kamp, 2017; Mueller et al., 2017; Van Kempen et al., 2018; WHO, 2018; European Environment Agency, 2020).

Methodologies for establishing the monetary cost of acute health impacts in general are relatively well developed as they are often used to demonstrate that health treatments are cost effective. These methodologies have more recently been expanded to include noise-related impacts that affect quality of life, but without acute health effects (eg, daytime annoyance and night-time sleep disturbance). Applying these methodologies, several countries and regions have developed estimates of the social cost of environmental noise. Some examples of these are provided in Table A.4. It is important to note that these estimates are not directly comparable as they employ different methodologies. More importantly, they cover different health impacts and the exposure-response relationships for some of these (and for hypertension in particular) are relatively uncertain.

Table A.4 Global estimates of the social cost of environmental noise

Study	Country / region	Noise source	Elements of social cost considered	Annual cost estimates
European Commission (1996)	European Union Member States	Environmental noise	Reduction of house prices, reduced possibilities of land use, increased medical costs, and the cost of lost productivity in the workplace due to illness caused by the effects of noise pollution	EUR13 million to EUR30 billion (NZ$23 million to NZ$53 billion)
European Commission (2011)	European Union Member States	Rail and road traffic	As above, and noting that 90% of total costs were related to passenger cars and goods vehicles	EUR40 billion (NZ$67 billion)
SWECO (2014)	Sweden	Rail, road, and aircraft traffic	Property price depreciation, illness due to noise exposure	SEK16.97 billion (NZ$2.86 billion)
UK Government Interdepartmental Group on Costs and Benefits Noise Subject Group (2010)	England	Environmental noise	Various health effects including annoyance	GBP7–10 billion (NZ$13–19 billion)
Hänninen et al. (2015)	Belgium, Finland, France, Germany, Italy, and the Netherlands	Road, rail, and aircraft traffic	Health	400–1,500 Disability Adjusted Life Years (DALYs)[22] per million people
Briggs et al. (2016)[23]	New Zealand	Road traffic	Health	821 years of life lost 917 healthy years of life lost

There have been no specific studies carried out in New Zealand to determine the cost of environmental noise; however, a study by Dravitzki et al., (2001) showed that exposure to traffic noise can result in more frequent property sales, and some of the early sales soon after construction of a road can be attributed to traffic noise. The Monetised Benefits and Costs Manual (Waka Kotahi, 2021) outlines an approach to valuing road traffic noise impacts based on the findings of international (predominantly hedonic price) valuations. Costs of road noise are valued at NZ$495 per year for each dB change across the total number of

[22] See section 0 for an explanation of DALYs

[23] The authors do, however, advise caution in using the findings as they are based on simplistic models of exposure and only consider exposure in the five main cities. Additional uncertainties are also likely in transferring the exposure-response functions from Europe to New Zealand, because of differences in environmental conditions and population characteristics; and no allowance was made for possible overlap or interaction between noise and traffic-related air pollution. For these reasons, the authors have low confidence in their estimates of the disease burden from noise.

households affected, where the costs per year are equivalent to 1.2% of the value of properties affected and an average value for urban property of NZ$640,000.

A.3.3 Structure of cost model literature review

The cost model literature review sets out the findings on approaches to valuing the health and wider social and economic impacts associated with exposure to transport noise. It should be read alongside the review of health impacts which examines the evidence on the relationships between exposure to noise from various transport sources and health impacts.

The remainder of this section is structured as follows:

- Section A.3.4 contains an overview of the approach used to undertake the literature review, including the scope of the review and the key research questions to be addressed.
- Section A.3.5 provides a high-level overview of the literature that was accessed.
- Section A.3.6 provides an overview of the components of social cost (health and non-health impacts) that will need to be considered in the social cost model.
- Section A.3.7 provides a summary of the review findings in relation to approaches used to quantify and value the burden of disease (ie, the health impacts).
- Section A.3.8 examines the approaches, methodologies, and data sources that have been used for valuing the direct and indirect costs of illness, for example, costs of hospitalisation, treatment costs, and ongoing care.
- Section A.3.9 describes the state of the art with respect to measuring and valuing productivity losses that can be associated with poor sleep or ill-health.
- Section A.3.10 examines the approaches and methodologies that have been applied to valuing the impacts of transport noise exposure on amenity.
- Finally, Section A.3.11 sets out conclusions and recommendations for the approaches to be used in the social cost model.

A.3.4 Approach

The main objective of this review was to identify best practice approaches and methodologies that may be used to inform the development of a robust yet practical approach to the economic valuation of transport-related health, productivity, and amenity effects in the social cost model. While there is now a substantive body of epidemiological literature around the effects of transport noise on human health (much of it summarised in WHO (2018) and van Kamp et al. (2020)), the literature on the economic valuation of health and amenity effects arising from transport noise is far less extensive. A systematic review of studies was undertaken to identify approaches used elsewhere and to evaluate their applicability to the New Zealand context.

Scope

In light of the above, the review is intended to provide a summary of the approaches that have been applied to valuing the social costs of prolonged exposure to transportation noise, where social costs include both health and non-health impacts.

It considers:

- conceptual approaches to quantifying and valuing health impacts (mortality and morbidity) and the relative strengths and drawbacks of these in terms of the validity or reliability of the estimates
- approaches that have been developed elsewhere and are applied in practice to value the social costs of environmental noise, mostly notably in the form of transport appraisal guidance and regulatory impact assessment

- approaches adopted for valuing the health impacts of other environmental externalities, and specifically air pollution, given that research in this area tends to be larger in volume and the approaches are arguably more established.

Note that the review is focused on understanding the broad approaches to quantifying and valuing the social costs of transport noise. Once the health impact review has been completed, it may be necessary to revisit the evidence collated for the purposes of this review in order to confirm the approaches to quantifying and valuing the specific health outcomes (eg, sleep disturbance, ischaemic heart disease, annoyance, etc) to be included in the social cost of noise model.

The review protocol, including the scope and search strategy, is provided in section A.3.1.

Review questions

The key research questions addressed in this review are as follows:

- What are the components of social cost associated with prolonged exposure to transport noise?
- What models, tools, and methodologies have been developed and used elsewhere for quantifying and valuing the burden of disease (eg, premature mortality and morbidity) arising from transport noise, what are their relative strengths and weaknesses, and how applicable are they to the New Zealand context?
- What approaches, methodologies, and data sources have been used for valuing the direct and indirect costs of illness (eg, hospitalisation, treatment costs, household assistance, etc)?
- What approaches, methodologies, and data sources have been used for measuring and valuing productivity losses? And to what extent can these be combined with approaches to valuing the burden of disease (eg, using value of a statistical life) without the risk of double-counting?
- What approaches, methodologies, and tools have been applied for valuing the impacts of transport noise exposure on amenity? And to what extent, if any, might these approaches or the values they generate result in double-counting?

A.3.5 Overview of the literature reviewed

Using the approach set out in the protocol, over 200 studies were identified as potentially relevant, of which around 70 were considered suitable for more detailed review and analysis. Although academic studies discussing methodological approaches were considered, the focus was on studies that described practical attempts to quantify and value, in economic terms, health and productivity effects, overcoming limitations in data and methods, with the aim of developing a pragmatic and practicable approach. Studies that focused on non-transportation noise were excluded, as were those that focused on the costs of structural measures for noise mitigation, and those of poor quality[24].

The studies reviewed were published between 2006 and 2021 and covered a range of geographies, including Canada, the United States, Europe, the UK, Australia, New Zealand, and South Korea. Over half (54%) of the studies reviewed had a focus on Europe or the UK. Around a quarter of the studies reviewed were case studies (at neighbourhood, city, or national scale), while the remaining studies took the form of either guidance and methodologies (37%) or evidence reviews (38%).

The majority of studies that quantify and/or value the health effects associated with prolonged exposure to harmful levels of transport noise typically follow an impact pathway approach. Such approaches provide a structured and transparent way of linking the sequence of events between changes in noise levels (eg, as a

[24] For example, a study published in the International Journal of Economics and Finance by Magablih (2019) on the economic cost of noise pollution in Jordan which cites evidence from the 1970s and 80s and contains various errors and omissions.

result of a transport-related intervention) and the outcomes or impacts that can be valued in monetary terms. A smaller number of the studies reviewed rely on damage costs (ie, the costs associated with each 1 dB increase or decrease in noise within a given range) although these unit costs are themselves typically derived by means of bottom-up impact pathway approaches (see section A.3.7).

Most of the studies reviewed calculate transport-related noise costs by combining estimates of the number of people or households exposed to harmful levels of noise, the proportion of these that go on to develop health effects, and the total health costs per case. The studies do, however, vary in the calculation of total health costs. The key differences include the following:

- **The dispersion models considered**. To calculate total noise costs, the number of people exposed to different levels of noise has to be known, which depends on the emission and dispersion of noise. Noise emissions are estimated based on the characteristics of the transport type being considered (eg, road traffic, rail, or aviation). There are several models available and widely used for estimating the dispersion of noise. Pertinent choices for this study are covered in the acoustics review.
- The **threshold** used for determining the point (noise level) at which noise has a discernible effect on health or amenity and therefore causes costs. Given that an increase from 50 to 55 dBA might cause at least a doubling of noise costs because the decibel scale is logarithmic, a threshold difference of 5 or 10 dBA is important for the total health cost figures. Different thresholds also apply for different health effects. The evidence around exposure-response relationships is examined in detail in the health review and so is not discussed further in this report.
- **Types of costs considered**. The majority of studies only considered the wellbeing costs of the individuals affected; others looked at the direct economic costs in terms of lost productivity, and others looked at the total costs (economic costs + medical costs + individual wellbeing costs) (see section A.3.6).
- **Health effects considered**. The economic literature often distinguishes between two main types of health effects: annoyance or amenity and other health effects such as sleep disturbance, cardiovascular diseases, and metabolic effects. However, not all studies consider both types. Furthermore, some studies and methodologies consider or recommend treating annoyance as a health effect (van Kamp et al., 2020; Van Kamp et al., 2018; DEFRA, 2014; WHO, 2018; Berry & Sanchez, 2014; Berry & Flindell, 2009), while others (Irwin & Livy, 2021; Kim et al., 2019; Fryd et al., 2017; Pignier, 2015; Austroads, 2014; CE-Delft et al., 2011) treat it as a disamenity effect valued using revealed (hedonic price models) or stated preference (contingent valuation) approaches (see sections A.3.7 and A.3.9). It is important to identify which effects are covered to avoid the risk of double-counting. For example, studies that only value annoyance may use hedonic pricing analyses that examine the impact of households' exposure to noise on property prices, but if these hedonic pricing approaches are combined with different approaches for valuing sleep disturbance, there is a significant risk of double-counting (see section A.3.10).
- **Approaches to valuing the burden of disease**. The studies are mixed in the choice of metrics used to measure the environmental burden of disease and the approach to valuing that burden in monetary terms. Most of the studies apply the concept of Disability-Adjusted Life Years (DALYs) following the lead of the WHO Guidance published in 2012, while others use, or recommend the use of Quality-Adjusted Life Years (QALYs). Similarly, while some studies have applied a centrally determined value of a statistical life (VoSL), others, and more frequently so, use a value of a statistical life year (VoLY) to value health impacts (see section A.3.7).

The following sections present the findings of the literature review with respect to each of the key research questions identified in section A.3.4.

A.3.6 The components of social cost associated with prolonged exposure to transport noise

The earliest studies on the valuation of noise impacts tended to centre on amenity impacts, or annoyance[25] (Navrud, 2002). Until fairly recently this was the approach used in the UK Department of Transport's Transport Appraisal Guidance (TAG), using values derived from hedonic pricing analyses that examine the impact on property prices of households' exposure to transportation noise. However, with growing interest in the health effects of environmental noise (see for example, WHO, 2011; Defra, 2014; European Commission, 2020; Recio et al., 2016; WHO, 2018; van Kamp et al., 2018; Brown & van Kamp, 2017), efforts are increasingly being made to incorporate health effects and related costs into values used in the appraisal of transport schemes. For example, the values used in Sweden are based on local hedonic pricing (HP) studies with the addition of 'a 42% mark-up… made to capture the value of 'un-conscious' health effects, ie, the effects of noise on residents' health that they are not aware of and hence are not reflected in house prices' (Eliasson, 2013, p. 6).

Hunt (2001), in a European Commission project to develop uniform accounts and marginal costs for transport efficiency ('the UNITE project'), provided the methodological basis for the economic valuation of endpoints. His starting point was the identification of the components that comprise changes in welfare. These included:

- **resource costs** – the direct medical and non-medical costs associated with treatment for the adverse health impact; that is, all the expenses the individual faces with visiting a doctor, ambulance, buying medicines and other treatments, or in undertaking defensive measures, plus any related non-medical cost such as the cost of childcare and housekeeping due to the incapacitation of the affected person
- **opportunity costs** – the costs associated with the loss of productivity (absenteeism or reduced capacity to perform while at work) and/or leisure time, including non-paid work, due to the health impact. It is the time spent with the illness as well as the time spent recovering. In general, it is measured as individuals forgone income due to the disease or death
- **disutility** – other social and economic costs including any restrictions on, or reduced enjoyment of desired leisure activities, discomfort or inconvenience (pain or suffering), anxiety about the future, and concern and inconvenience to family members and others. Unlike resource and opportunity costs which have established market prices, impacts on wellbeing cannot be observed in markets and therefore require the use of specific techniques such as willingness to pay (WTP) surveys for eliciting individuals' preferences regarding their disutility costs.

Similarly, Markandya et al*.* (2019) note that impaired health impacts individuals' wellbeing in several ways: via increased costs for treating the illness; losses of income due to impediments to work; and losses of utility related to the pain, suffering, or anxiety associated with the illness. In the extreme case when the health impact is death, individuals suffer total loss of wellbeing. In both cases, wellbeing may also be impacted via the costs of potential defensive measures (eg, noise insulation) taken to avoid illnesses or to maintain a health status. Markandya et al. (2019) adopt the same typology of costs as Hunt (2001).

This is not dissimilar to an earlier study on the health costs of air pollution (Sommer et al., 1999) which characterises the health costs in terms of:

- **mortality** – loss of life

[25] Amenity impacts are defined by the UK Interdepartmental Group on Costs and Benefits Noise Sub-Group as the conscious annoyance or negative reaction to noise exposure.

- **costs of illness** – the loss of production due to incapacity to work and the medical treatment costs. These costs determine the 'material part' of the health costs. These are assessed on the basis of market prices (loss of earnings, costs of medication, costs per day in hospital, etc) and may be both individually and collectively borne depending on the nature of the healthcare systems in operation and the extent to which the individual is responsible for covering the costs of his/her own care
- **costs of averting behaviour** – those costs which result from a different behaviour due to air pollution. Examples of these may include abstention from practising outdoor sport activities during a summer day with a high ozone concentration, the installation of air filters, or a different choice of residential location due to air pollution (eg, moving out of inner cities). These are treated as a component of resource costs in Hunt (2001) and Markandya et al. (2019)
- **intangible costs** – these reflect the individual loss of utility and consist of the pain, grief and suffering due to an illness and are thus the same as the disutility costs identified by Hunt (2001) and Markandya et al. (2019).

The authors do, however, caution against the risk of double-counting by combining all three components of costs. This is echoed by several others (eg, Navrud, 2002; WHO, 2008) and is related to the fact that intangible costs are typically elicited by asking a representative sample of respondents in a population how much they would be willing to pay to avoid or reduce the risk of a particular health outcome. Respondents may factor in the financial costs (ie, costs of illness and costs of averting behaviour) into their response thereby introducing a degree of overlap between components. Financial costs are, however, often not borne fully by the individual but are shared through health insurance and public health care provision. If it is not possible to separate out the private (ie, costs borne by the individual) and social costs (ie, costs borne by the public), then a part of the disutility measured in the WTP estimate will be incorporated in the private medical costs associated with treatment (or prevention) of the health endpoint, and the total valuation should therefore be reduced by an equivalent amount.

WHO (2008) suggests two ways in which this issue may be addressed. The first is to calculate net economic production losses instead of gross economic production losses. This means that the lost (future) consumption is subtracted from the total economic production losses and is therefore excluded from the cost-of-illness valuation. The second is to exclude the cost-of-illness costs borne by individuals.

The UK Department of Health's guide for quantifying the health impacts of government policies (2010) identifies two broad areas of cost to be considered: resource costs and life years lost. It defines resource costs as the health care costs (ie, the costs associated with treating patients), costs incurred by local authority services such as Social Services, and costs to patients and their families (eg, care at home, medication).

A study conducted on behalf of Public Health England (Pimpin et al., 2018) on the impacts of air pollution distinguished five different categories of health and healthcare-related costs associated with the impacts of air pollution. The types of costs covered were:

- **primary care** – the primary point of contact of someone seeking care. GP visits are the main source, but the costs may also include nurse visits, home visits, and consultations by telephone or email
- **prescription costs** – usually estimated as the volume times the costs of primary care prescription
- **inpatient costs** – the total costs of treating a patient at hospital for a specific diagnosis (episode). They include day cases, elective, and emergency admissions
- **outpatient costs** – the costs of visits to specialists
- the costs of **social care** – these may encompass a broad range of activities associated with the tasks of everyday living, from child protection services to end-of-life care. The broad coverage, lack of a clear

definition, lack of data, and need to rely on proxy measures meant that the authors were unable to quantify the costs in the health impact assessment.

The updated Health and Air Pollution in New Zealand (HAPINZ 2.0) study (Emission Impossible et al., 2012) identifies three main components of social costs of pollution-related health effects:

- **Loss of life and life quality** – valued in terms of the official VoSL in New Zealand for loss of life.
- **Costs of medical treatment** – valued in terms of the costs of hospital admissions for the average length of stay associated with each of the relevant health endpoints.
- **Loss of output** – measured in terms of the loss of output during hospitalisation and the number of restricted activity days (RAD)[26]. Output is valued on the basis of the average wage, salary, or income per day per person regardless of employment status, age, or day of the week (ie, it includes non-working days).

At the time this review was undertaken, work on an update to the HAPINZ 2.0 study was underway. The findings and recommendations of the final methodology for estimating social costs will need to be taken into account in the development of the SCON approach so that the approaches can be aligned as closely as possible.

The UK Department for Environmental, Food and Rural Affairs (DEFRA, 2014) recognises four broad groups of impacts, based on a general framework developed by the Interdepartmental Group on Costs and Benefits Noise Sub-Group (IGCB (N)) in 2008. These are:

- **health**, including both morbidity and mortality
- **effects on amenity**, which reflects individuals' conscious annoyance from noise exposure
- **productivity**, which relates to impaired economic performance as a result of noise-related sleep disturbance or noise acting as a distraction
- **environment**, where noise may impact on the functioning of ecosystems, such as through bird breeding patterns (Francis et al., 2009).

At present only the health and annoyance costs are included in UK transport appraisal. While investigative research into the impact of noise on productivity has been undertaken (Morgan et al., 2011; URS, 2014), there remain substantial gaps in the evidence base.

Similarly, there is limited research on the economic impacts of transportation noise on wildlife and, where research has been conducted, the evidence is mixed or cannot easily be extrapolated from individual or group level to population-level effects (IGCB(N), 2008; EEA, 2020). While there are increasingly sophisticated ways of valuing biodiversity, ecosystem services and even individual species, less is known about the relationships between noise (type and level) and behavioural responses in wildlife. Impacts on ecosystem functioning have therefore not been considered further in this review.

A.3.7 Approaches to quantifying and valuing the burden of disease from transport noise

This section focuses specifically on the findings of the review in relation to quantifying and valuing the health impacts (mortality and morbidity, or disutility costs) associated with transport noise. The specific health endpoints or outcomes to be valued in the model will be determined through a separate review of the health

[26] This is when, for example, air pollution exposure causes symptoms sufficient to prevent usual activities such as attendance at work or study, but not sufficient to seek medical attention.

literature conducted in parallel with this review. The focus here is therefore on the broad approaches that have been applied to converting health outcomes into monetary values.

Quantifying health impacts

Environmental health impacts are evaluated separately for morbidity and mortality. For example, for mortality the health endpoint is death, which is a discrete, well-defined and observable event. However, non-fatal health effects can vary in severity and length of duration of illnesses, and are subjective, varying with individuals' perceptions of the associated symptoms. This in effect means that the individual cost of morbidity may vary from one individual to the next depending on factors such as the number of days with symptoms or restricted activities, to number of days in a hospital, or life-long effects of chronic diseases (Markandya et al., 2019).

The health effects of transport noise involve not only mortality and morbidity but also aspects of the quality of life such as the aggravation of pre-existing disease symptoms and severe annoyance. Integrating such a diverse range of health effects is a challenging task. One method that facilitates the aggregation of different health effects is the use of standardised measures of population health such as DALYs or QALYs (see Table A.5). Health effects can also be expressed in monetary terms. This requires the task of expressing loss of life, life-years, or burden of disease in monetary units.

Table A.5 DALYs and QALYs

DALYs and QALYs
DALYs indicate the estimated number of healthy life years lost in a population from premature mortality or morbidity (ie, the health burden). The DALY is calculated as the sum of years of potential life lost due to premature mortality and the years of productive life lost due to disability. By combining mortality and morbidity effects into a single measure, the DALY facilitates assessment of the health effects of different health outcomes in a comprehensive and comparable way. It can be calculated as follows: $$\text{DALY} = \text{YLL} + \text{YLD}$$ Where: YLL = ND (number of deaths) x DW (disability weight) x LD (standard life expectancy at age of death in years) YLD = NI (number of incident cases) x DW (disability weight) x LI (average duration of disability in years) Since the 1990s the WHO and the Institute for Health Metrics and Evaluation (IHME) have produced numerous global burden of disease (GBD) estimates, using DALYs. For each country, age- and sex-specific estimates of the number of YLD and the YLL for different diseases are available for download. Burden of disease estimates are also sometimes available at national and local level. As part of a recent review by O'Donovan et al. (2018) 198 studies were identified that produced DALY estimates for specific populations or geographies within Europe. In contrast, a QALY is a measure of the state of health of a person or group in which the benefits, in terms of length of life, are adjusted to reflect the quality of life. One QALY is equal to one year of life in perfect health. QALYs are calculated by estimating the years of life remaining for a patient following a particular treatment or intervention and weighting each year with a quality-of-life score (on a 0 to 1 scale). It is often measured in terms of the person's ability to carry out the activities of daily life, and freedom from pain and mental disturbance. The two concepts are not equivalent. DALY measures adverse impacts while QALY measures good health. For QALYs, perfect health is assigned a score equal to one and while score of zero represents death. For DALYs, the opposite applies. QALY and DALY estimates might differ for the same illness as they are based on individual preferences and expert estimates respectively. Different techniques to elicit QALY could also produce different results, but both QALY and non-market valuation techniques (ie, revealed and stated preference approaches) are based on individual preferences.

Broadly speaking, there are two main approaches to quantifying and valuing the health impacts of noise: the ‘impact pathway’ approach, and the ‘damage cost’ approach. The impact pathway approach involves a ‘bottom up’ calculation in which environmental benefits and costs are estimated through a chain of steps that link the value of health impacts to the size of the affected population and ultimately to the type and level of emissions produced. The overall impacts are calculated using the following general relationships:

- Impact = quantity of emissions x size of the exposed population x response function
- Cost = Impact x unit cost of impact

Applying the impact pathway approach to every policy impact assessment is very resource-intensive, particularly when there are several health endpoints to consider. As a result, many countries have adopted tables or models to allow direct valuation based on emissions levels alone. These are frequently referred to as ‘damage costs’, stated as a cost per decibel change in noise. Damage costs for a specific country or jurisdiction are, however, often generated via a full impact pathway approach, utilising location-specific inputs and data, but are sometimes drawn from studies from elsewhere using a technique called benefits (or value) transfer. These are typically used for the purposes of evaluating the health costs of transport interventions (eg, road or rail improvements or airport expansion). This is indeed the approach used in New Zealand, which draws on international studies to recommend a value per household per year per dB noise increase (Waka Kotahi, 2021).

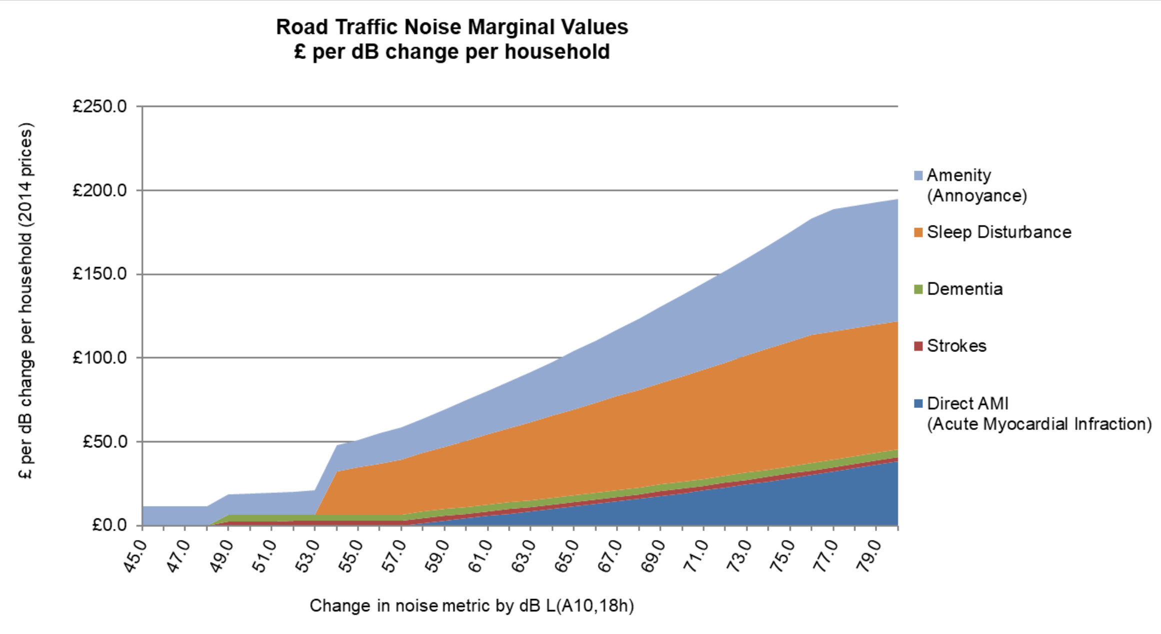
While different countries have adopted different approaches (Mackie & Worsley, 2013), many often rely on the same underlying health studies. The most advanced and detailed studies are arguably those undertaken in the UK and Europe, where independent scientific committees have provided advice on health quantification and valuation.

In England, the current guidance for valuing impacts associated with environmental noise is published by DEFRA on behalf of the IGCB(N). The IGCB is a DEFRA-led group of government analysts and policy officials that provides analysis and advice relating to the quantification and valuation of local environmental impacts. It aims to do this through the development of methodologies, evidence reviews, and subsequent recommendations.

The IGCB(N) set out its recommended approach for valuing health impacts associated with environmental noise in two documents in 2010 and 2014. The relevant health outcomes are annoyance, sleep disturbance, stroke and dementia (via hypertension), and acute myocardial infarction. The approach proposed by the IGCB(N) uses exposure-response functions to estimate the prevalence of these health impacts in the population, which is then monetised in relation to the value of a QALY. Once the additional number of noise-related disease cases is calculated, a monetary value can be estimated.

These recommendations have subsequently been incorporated into the government’s Transport Appraisal Guidance (UK Department for Transport, 2019) as well as central government guidance on appraisal and evaluation of policies, projects and programmes (HMT, 2020). The Transport Appraisal Guidance provides marginal values (GBP(£) per dB change per household) for each health endpoint and noise source as illustrated in Figure A.2 for road traffic noise.

Figure A.2 Road traffic noise marginal values (DEFRA, 2014)



The effects of environmental noise are currently valued using a DALY framework. This is based on a value assigned to a life year lived in perfect health and a set of adjustment factors – known as disability weights – that adjust the perfect health value for different conditions in relation to their severity of impacts on an individual's quality of life. It considers both the impact on length of life (longevity) and health-related quality of life.

The approach used in the UK is broadly consistent with that set out in the WHO (2012) guidance on estimating the burden of disease from environmental noise and the more recent 'Environmental Noise Guidelines for the European Region' (WHO, 2018). The IGCB(N) is currently considering whether it needs to update the approach to valuing the health impacts of environmental noise in light of more recent evidence. In particular, it is investigating the strength of the evidence and possible approaches to assessing and valuing the impacts of transport noise on both productivity and a number of additional health endpoints, including metabolic diseases (stroke, diabetes, obesity), cognitive effects in children, mental health (excluding annoyance, but including wellbeing and quality of life), reproductive health, cancers, and cognitive degeneration (Notley et al., 2019; van Kamp et al., 2020).

Valuing premature loss of life (mortality)

The most commonly used metric for representing the value of a life is the VoSL. The VoSL aims to reflect the amount of money that an individual or society is willing to pay to save one human life. By convention, the life is assumed to be the life of a young adult with at least 40 years of life ahead (Australian Government, 2019). It is a statistical life because it is not the life of any particular person.

Based on the literature reviewed, it is possible to distinguish three general approaches to valuing mortality, which are:

- the human capital approach (also known as gross production or consumption loss)
- willingness to pay, VoSL, or value of preventing a statistical fatality (VPF)
- value of a statistical life year (VoLY).

The **human capital approach** estimates the economic productivity of an individual as equal to the discounted lifetime earnings. This assumes that the value of an individual life can be approximated by what this individual produces for society, and that individuals' earnings approximate their productivity (Markandya et al., 2019; Social Value UK, 2016). It has been widely used for its ease of use but important criticisms of the approach (Social Value UK, 2016) are that it:

- places a much lower value on saving children's lives compared to saving the lives of adults because of discounting and the time lag before children become productive participants in the labour market
- tends to value the health of women and non-white individuals less than the health of adult white males because of observed earning differences among individuals of different gender and race in many countries
- assigns a VoSL of near zero to groups with low potential earnings as the long-term unemployed, retired, or those with severe disabilities, with no regard to the value they place on their own lives
- lacks the ability to quantify the intangible elements, such as leisure and social interaction, which are hard to value but are nevertheless important.

As this method does not consider the intangible factors which dictate the value people place on their lives, it can only be considered a basic tool for determining VoSL.

In contrast, the willingness to pay **WTP approach** assumes that the preferences of individuals can be characterised by the substitution between income and a particular health status; that is, individuals make trade-offs between the consumption of goods or services and factors that improve their health conditions and/or can save their lives. It has its basis in the premise that changes in individuals' welfare can be valued according to what they are willing and able to pay to achieve that change.

Similarly, the willingness to accept (WTA) measure can be defined as the minimum amount of money the individual would require to forgo some good or service, or to bear some harm. Thus, individuals treat their health like any other good and reveal their preferences through the choices that involve changes in the risk of death or injury and in the consumption of other goods (eg, defensive expenditure) whose values can be measured in monetary terms. Note that WTP measures may include all components of the social cost of noise described in section A.3.6.

For the valuation of mortality, WTP estimates represent the income individuals would give up to reduce their risks of death or to increase their life expectancy. The aggregate WTP for a measure saving a number of lives, divided by the number of lives saved, defines the value of a statistical life. An alternative way of measuring the value of changes in risk of death is in terms of life expectancy, in which individuals express their willingness to pay for a small change in life expectancy, which can then be reported in terms of a value of a life year, or VoLY.

There are three basic non-market valuation approaches suggested for identifying the WTP of an individual for mortality risks (OECD, 2010; Social Value, 2016; Australian Government, 2019):

- **The hedonic wage or labour market method** which examines wage differentials for jobs which carry different risk of injury or death (an example of the revealed preference method). Wage differentials occur for many reasons. A job may have better working conditions, more flexible working hours or opportunities for career advancement. Some of this differential can be explained by differing analyses actual behaviour in the labour market. If a person is working in a job with above-average mortality risk, he or she will normally require a higher wage to compensate for this risk. By observing the wage premium, one can see what value they attach to that risk. This approach is, however, not well-suited to the valuation of health impacts from environmental noise.

- **The contingent valuation or stated preference (SP) method** in which studies explicitly ask individuals how much they would be willing to pay (or willing to accept) to compensate for a small reduction (increase) in risk. The main appeal of SP methods is that, in principle, they can elicit WTP from a broad segment of the population and can value causes of death that are specific to environmental risks. The main drawback of the SP method is that it is hypothetical, so that the amounts people say they are willing to pay may be different from what they actually would have been willing to pay, if faced with the given situation.
- **The consumer preference method** which works in much the same way as a worker's WTP for a safer work environment might reveal the value they place on their own life. Consumer decisions where health risks are a factor can also be used to infer VoSL. By way of example, if a consumer is willing to pay NZ$5,000 for double glazing which decreases their likelihood of death by 0.05% then it could be said they place on their life a value of NZ$10 million (NZ$5,000 x (1/0.0005)).

A common criticism of the concept of a VoSL is that the life expectancy of an individual is often not taken into account. Under the WTP framework, the VoSL might be expected to decline with age: as a person gets older, they have less of their life remaining. Studies have shown, however, that the VoSL is influenced across the life cycle by an individual's family and employment circumstances, and risk aversion may also vary with age. Research suggests that an adult's WTP increases to around age 40, before declining (Aldy & Viscusi, 2007). To address this, the concept of a **value of a life year** (VoLY) has emerged as a more meaningful estimate. The VoLY aims to calculate the value of one additional year of life experienced and is often used in cost-benefit analyses regarding healthcare procedures.

Common criticisms of the WTP approach more generally (Sommer et al., 1999) are that it:

- depends on respondents' income levels: individuals with a high income are able to spend more money on their health and wellbeing than people with a low income. This income-related valuation of the value of preventing a statistical fatality may be ethically problematic, especially when it applies to very different countries
- may neglect some of the indirect costs of illness, particularly productivity losses, in cases where these are not borne by the individual (for example, if the corresponding loss of income is totally remunerated by the social insurance system) even though society as a whole would be affected by the productivity losses
- may over- or underestimate the value of individuals' wellbeing. Respondents may not understand the levels of risk at stake, or the consequences on their health. It is also difficult for individuals to be familiar with small variations of risk. This may imply large discrepancies between individual valuations.

But the main difficulty of the WTP approach consists of obtaining reliable and correct empirical estimations. A multitude of empirical assessments conducted so far for the value of statistical life have provided a very large range.

There is no consensus as to the most appropriate approach and, in many instances, different approaches may be more, or less, appropriate depending on the specific nature of the health impacts. For example, Markandya et al. (2019) suggests that VoSL may be more appropriate for valuing premature mortality associated with acute impacts (ie, where death is attributable to short-term exposure, for example from severe air pollution) whereas a VoLY, which considers the life expectancy over the whole population, is more appropriate for chronic impacts (ie, where prolonged exposure may result in death, but is not directly attributable). Similarly, in their report for the Australian Commonwealth Department of Environment, Water, Heritage and the Arts (DEWHA), Jalaludin et al. (2009) recommend that the use of the VoLY is preferable to the use of the VoSL in monetising the air pollution effects on premature mortality and should be used whenever feasible and practicable.

Following a review of research into VoSL and VoLY and of international guidelines for life and health values, Abelson (2007) suggested public agencies in Australia adopt a VoSL of $3.5 million (NZ$3.99 million), a constant VoLY of $151,000 (NZ$172,140) which is independent of age, and age-specific VoSLs for older persons equal to the present value of future VoLYs of $151,000 (NZ$172,140) discounted by a private time preference discount rate[27] of 3 percent per annum. Each of these is measured in 2007 dollars.

The Australian Government recommends the use of estimates of VoSL and VoLY derived from previous estimates with appropriate adjustments to account for inflation. For morbidity, it suggests that one method to value these benefits is to adjust the VoLY (which could be interpreted as the value of a year of life free of injury, disease and disability) by a factor that accounts for the type of injury, disease or disability. The Australian Institute of Health and Welfare has published disability weights for most diseases and injuries that can be used to adjust the VoLY.

Loss of quality of life (morbidity)

The term ‘Quality-Adjusted Life Year’ (QALY) was first employed in the 1970s to indicate a health outcome measurement unit that combines duration and quality of life into a single unit (Zeckhauser & Shepard, 1976, cited in Sassi, 2006). By the mid-1990s the QALY framework was widely accepted and used as the reference standard in health economic evaluation despite continuing debate on its theoretical underpinnings and practical implications (Sassi, 2006).

The key idea underpinning the QALY is that people are not exclusively focused on the extension of remaining life expectancy as the sole measure of the value of a health care intervention but are also concerned with the quality of their health during their lives; and indeed, may be willing to ‘trade’ between the two. For example, people may be willing to undergo unpleasant treatments and endure periods of reduced quality in order to increase their life expectancy; but they may also prefer to forego some debilitating or disfiguring intervention in order to have a higher quality of life, albeit for a shorter period of time (Loomis, 2009, in Berry & Flindell, 2009).

Most of the challenges to the QALY framework have been based on the difficulties involved in making interpersonal comparisons and aggregating individual utilities; the assumptions on which health utility elicitation methods are based; and the implicit discrimination against the elderly and the chronically ill or disabled (Sassi, 2006).

Baker et al. (2010) argue that suitable existing evidence is scant and of variable quality. Some estimates have been made of the value of a QALY based either on modelling approaches or on survey research. Moreover, survey work on the value of a QALY has been limited. Typically, individuals have been asked about their WTP for health gains for which quality adjustment factors have been obtained from another sample without fully adjusting for uncertainty (ie, by presenting scenarios involving certain gains in quality of life) and, in some cases, eliciting values from patients and not from members of the general public.

Disability-Adjusted Life Years

The QALY framework provided a basis for the development of a number of other health outcome measures, including the Disability-Adjusted Life Year (DALY) which was developed in the early 1990s (Sassi, 2006) by the World Bank and soon thereafter was adopted by the WHO to calculate the global burden of disease. Since then, it has been used to estimate the cost of environmental hazards on health, including noise pollution.

[27] This is the rate at which individuals are willing to forgo an additional unit of consumption now for an additional unit of consumption in the future. It is to be distinguished from the more commonly applied social time preference rate which is the value that society places on present consumption relative to future consumption.

The DALY estimates how much disease affects the life of the population by combining the burden from:

- mortality, in terms of years lost because of premature death due to disease
- morbidity, in terms of years of life lived adversely affected by disease.

There are several ways to calculate the burden of disease (van Kamp et al., 2018). In most cases the population attributable fraction (PAF)[28] is used. This is the proportion of people with a disease which can be attributed to an environmental exposure. In the next step, the number of people with a disease due to a given environmental exposure multiplied by the average duration of this disease and the weight for the severity of this disease is calculated. This weight ranges from 0 to 1 (death). DALYs tend to overestimate the effects as, for example, people who are sleep disturbed may also be annoyed by noise and run increased risk for long-term health effects.

As noted above, one DALY corresponds to one lost year of healthy life, attributable to morbidity, mortality, or both (see Table A.5). The sum of DALYs across a population provides a measurement of the gap between actual health status and an ideal situation in which the entire population lives to an advanced age, free of disease and disability. As a measure of outcome in economic evaluation, the DALY differs from the QALY in a number of aspects. These include the following (Markandya, 2019):

- DALYs are framed as years lost from a global ideal length and quality of life, whereas QALYs lost are framed against the QALYs that would have been achieved by the population of interest.
- DALYs are weighted in favour of adults in view of the fact that they support children and the elderly, whilst QALYs account for age in the average healthy utility value of the population.

Finally, the weights applied to the years lived differ in DALYs and QALYs; DALYs use disability weights, a measure of the severity of disease bounded between 0 for full health and 1 for death. It is not possible to exist in a state worse than death. Disability weights do not reflect the wider quality of life impacts of disease, such as impact on daily activities and mental health. The method of valuing severity also differs for disability weights. Generally, a 'person trade-off approach' is used and this has typically been based on the opinion of experts rather than patients and the public. Such values are therefore not compatible with the estimation of QALYs which require a preference-based valuation of generalised health-related quality of life.

Most importantly, the DALY incorporates an age-weighting function assigning different weights to life years lived at different ages (Sassi, 2006). Key challenges to the DALY framework have focused on the equity implications of age-weighting and on the methods used to assess disability weights (Sassi, 2006).

A.3.8 Approaches to valuing the financial (direct and indirect) costs of illness

As noted in section A.3.6, the financial costs of illness relate to:

- the costs associated with visits to medical practitioners
- the costs of hospitalisation
- prescription or self-prescribed treatment costs
- the costs of ongoing rehabilitation and homecare.

Few of the studies and appraisal methodologies reviewed include the direct and indirect costs of illness, possibly because previous work has shown that these are seldom significant in overall monetary terms when compared with health effects. The findings of those that did are summarised below.

[28] The contribution of a risk factor to a disease or a death is quantified using the population attributable fraction (PAF). PAF is the proportional reduction in population disease or mortality that would occur if exposure to a risk factor were reduced to an alternative ideal exposure scenario (eg, no exposure to harmful levels of noise).

The cost of hospital admissions and other morbidity outcomes are usually based on the average use of hospital or medicinal resources for a patient group. For example, the 2012 HAPINZ study (Emission Impossible et al., 2012) estimated the **average medical cost per hospitalisation** would be about $6,040 and $4,330 respectively for cardiovascular and respiratory diseases. This was based on estimates for average length of hospitalisation of 5 days for cardiovascular diseases and 3.3 days for respiratory diseases and costs per day. The cost per day of hospitalisation was taken from Ministry of Transport estimates in 2010.

Hillman et al. (2018) examined the financial costs of inadequate sleep and its secondary outcomes. The financial costs considered were partitioned into those pertaining to health care, informal care, non-medical costs of workplace and motor vehicle accidents, productivity losses, and deadweight loss from inefficiencies associated with forgone taxation revenue and welfare payments.

The **health system costs** considered were both those directly associated with sleep disorders and those associated with conditions attributable to inadequate sleep including workplace injuries and motor vehicle accidents, stroke and depression, and heart disease and diabetes. The proportions of these related conditions and their associated costs that were attributable to an underlying sleep problem were calculated using a standard PAF methodology. The health costs accounted for included all expenditure in the Australian health system for the care of sleep disorders and for the care of other inadequate sleep-associated health problems, including costs of hospital care, health practitioners, pharmaceuticals, diagnostic tests, health aids and appliances, aged care, research, community and public health, and capital and administration. These data were derived from the latest available Australian Institute of Health and Welfare data, adjusted where appropriate to the appropriate price year values using the health price index.

Informal care costs were estimated for time spent by carers in providing assistance and support to people with inadequate sleep-related health problems outside the formal healthcare sector. This time could be used for work activities or as leisure time. Thus, although the time is given free of charge, it has associated opportunity costs due to a loss of economic resources. The cost calculation assumed there would be no care requirements due to inadequate sleep itself, only to conditions attributed to inadequate sleep that increased personal, household, and other care needs. Costs were based on the following care requirements:

1. For motor vehicle accidents, the average care requirement was estimated to be 4.5 hours per week.
2. For workplace injuries, the average care requirement was estimated to be 3.7 hours per week.
3. For cardiovascular disease (cerebrovascular disease, coronary artery disease, and congestive heart failure), the average care requirement was estimated to be 1.1 hours per week.
4. For type 2 diabetes, the average care requirement was estimated to be 0.1 hour per week.

Lack of adequate data precluded calculation of informal care costs for depression. The hourly cost of informal care was based on Australian Bureau of Statistics average weekly earnings estimates by age and gender, which was adjusted to the relevant price year value using growth in average weekly earnings.

The PAFs for motor vehicle accidents and for workplace accidents were used to derive the **nonmedical costs** of each from their respective total costs. These costs included those related to legal expenses, costs of investigation, aids and modifications to the home, respite services, travel costs and delays, correctional services, vehicle unavailability and repairs, towing, insurance administration, non-vehicle property damage, and fire and emergency services. The unit costs were derived from an earlier study by Deloitte Access Economics (2009) inflated to current prices using the consumer price index.

Work undertaken on behalf of Public Health England (Pimpin et al., 2018) extracted data on **prescription costs** from the literature using PubMed and MeSH terms[29], or Google when no peer-reviewed articles reported the costs of interest. **Inpatient costs** were analysed using the Hospital Episodes Statistics (HES) data. The authors identified patients with the conditions of interest based on the main diagnosis at admission (using the International Classification of Diseases, ICD-10) and considered the total treatment an individual receives under the care of a single consultant for a particular health condition. Outpatient costs (the costs of visits to specialists) were extracted from the literature.

Other potential sources of information may include:

- national estimates of the ‘hotel’ costs of hospitalisation
- the WHO CHOICE (Choosing Interventions that are Cost-Effective) database which includes country-specific costs for inpatient and outpatient health service delivery. Although the costs are based on country datasets gathered in 2008–2010, a recent analysis by Stenberg et al. (2018) concludes that the values are statistically robust and suitable for use as inputs for economic analysis. These values would only be used in the absence of more up-to-date estimates for New Zealand.

The HAPINZ study is currently undergoing a further refresh with the final social cost methodology expected in mid-2021. These will be reviewed once available so that the valuation basis, including data sources and methodologies used for noise pollution are consistent with those used for air pollution where appropriate.

A.3.9 Approaches for measuring and valuing productivity losses

The productivity losses associated with noise, such as those caused by sleep disturbance, health effects, workplace distraction, and (in early life) diminished academic performance are not well researched in terms of monetisation. There is a particular gap in the published evidence of the impacts of transport noise on productivity via the sleep disturbance pathway. While there are studies that examine the impacts of transport noise on sleep, and studies that examine the effects of poor sleep on productivity, the metrics used in each case do not support the ability to draw strong conclusions from the literature on the relationship between sleep disturbance attributable to transport noise and next-day productivity at work (URS, 2014).

A 2003 Japanese study (cited in Muirhead et al., 2011) estimated that the productivity loss due to sleep disturbance for noise from all sources cost the Japanese economy US$30.7 billion per year. A more recent report in Australia (Deloitte Access Economics, 2017) found the productivity losses associated with inadequate sleep amounted to the equivalent of around NZ$20 billion (or NZ$2,800 per person with inadequate sleep) in 2016–2017.

Though these studies are useful at informing us about the scale of the impact of noise on productivity, there still appear to be gaps in the evidence base which have prevented a robust monetisation methodology being adopted by policymakers to date. A recent report by van Kamp et al. (2019), for example, describes the results of the first stage of an update of a literature review into the effects of environmental noise on annoyance, sleep disturbance, metabolic and cardiovascular effects in the period between 2015 and 2019, primarily aimed at the identification of new publications and selection of eligible studies. The new search revealed 22 new studies addressing the effects of noise on sleep. Following a review of these, the authors concluded that the studies provided inconclusive evidence on the nature of the relationship between transport noise and sleep disturbance effects and the outcome measures are not always comparable,

[29] MeSH (Medical Subject Headings) is the National Library of Medicine’s controlled vocabulary thesaurus, used for indexing articles for the PubMed database. Each article citation is associated with a set of MeSH terms that describe the content of the citation. PubMed is a search engine for accessing a database of references and abstracts on life sciences and biomedical topics.

possibly due to methodological differences. The authors do, however, highlight new evidence from studies in France (Nassur et al., 2017, 2019) and Germany (Penzel et al., 2017) on the relationship between aircraft noise and sleep disturbance as worthy of further investigation, particularly for use in Europe, and recommended a meta-analysis for self-reported sleep disturbance for all transport sources combined.

There are potentially many different ways in which noise pollution can lead to losses in productivity. Morgan et al. (2011) examined three broad areas in which noise may impact upon productivity, which include:

- during working hours, for example as a result of distraction, loss of concentration, and difficulties communicating
- outside of working hours, for example as a result of sleep disturbance, lack of relaxation time, or increased stress
- impaired academic performance as a result of sleep disturbance, loss of concentration, and communication difficulties resulting in later-in-life productivity losses.

The report concluded that the sleep disturbance pathway provided the strongest link amongst the three. As high levels of night noise cause sleep disturbance that can affect next day performance, it is reasonable to assume that someone who is sleep deprived is less productive than someone with a full night's sleep. The authors thus suggested that it would be possible to develop a methodology and derive a monetary value for the productivity losses associated with noise-related sleep disturbance in the UK.

Following this, DEFRA commissioned further research to bridge the evidence gap in appraising the productivity impacts of environmental noise for policy and programme appraisal by investigating the evidence available to develop an understanding of the relationship between noise exposure, sleep disturbance, and productivity, building on the work of Morgan et al. (2011). The sections that follow are based on the findings of a literature review conducted for that study, supplemented with information retrieved from more recent studies.

In addition to the factors above, productivity losses may also be incurred as a result of incapacity to work as a result of hospitalisation or illness associated with cardiovascular diseases or metabolic effects that may be exacerbated by prolonged exposure to noise (see section A.3.6).

Defining productivity

There are a number of definitions of productivity used within economic and policy analysis. At its simplest, productivity can be defined as the ratio between inputs (capital and labour) and outputs (goods and services) in the production process[30].

Input can be measured in different ways with the choice of input indicating the type of productivity measure. For example, labour productivity can be measured in terms of the number of jobs, workers, or hours worked[31] required to produce a given level of output or conversely, the amount of economic output produced by a unit of labour input (ONS, 2013).

Measures of output are typically reported in one of three different ways (see Table A.6):

- total output
- gross domestic product
- gross value added.

[30] See, for example, https://www.stats.govt.nz/topics/productivity

[31] Output per hour is the preferred measure of productivity as it takes into account changes in average hours worked by individuals in the economy.

Table A.6 Measures of output

Measures of output
Total output
Total output, or simply 'output', is a measure of the value of goods and services produced by a country, in a set time period. This value holds whether the goods and services produced are consumed or used for further production. Total output differs from other measures of national output as it is not a value-added measure but a measure of the gross value including the value of production and the value of the intermediate inputs.
Gross domestic product
Gross domestic product (GDP) is the market value of all final goods and services produced within a country in a year, or other given period of time. Thus, it is a measure of national output and a key indicator of the state of the whole economy, and is used to compare economic growth within and across regions over time. It is estimated by combining the total value of national outputs, subtracting direct taxes and adding any government subsidies. In New Zealand, two approaches are used to estimate GDP: 'production' and 'expenditure'[32].
Production approach: Also known as the value-added approach. This measures the total value of goods and services produced in New Zealand, after deducting the cost of goods and services used in the production process.
Expenditure approach: Also known as gross domestic expenditure, or GDE. This measures the final purchases of goods and services produced in New Zealand. Exports are added to domestic consumption, as they represent goods and services produced in New Zealand. Imports are subtracted, as they represent goods and services produced by other economies.
Gross value added
Gross value added (GVA) is compiled from the National Accounts and is a measure of the contribution to the national economy by each individual producer, industry, or sector in New Zealand. It is used in the estimation of GDP using the 'production approach' and is measured as the total value of output of goods and services produced less the intermediate consumption (goods and services used up in the production process in order to produce the output).
When using the production or income approaches, the contribution to the economy of each industry or sector is measured using GVA.

Given the relationship between inputs and outputs, national productivity can be increased (or reduced) in two main ways:

- through increasing (or lowering) the level of employment or hours worked, so that the total labour input (and hence output) in the economy increases (decreases)
- through increasing (or decreasing) the amount of output each person produces: that is, increasing (or decreasing) their productivity.

Noise-related sleep disturbance and productivity

In relation to sleep disturbance, there are few published studies on the relationship between inadequate sleep and productivity that include a formal definition of productivity that resembles that outlined above. Rather, for many of the studies, productivity is implied by the particular characteristics that were analysed and include:

- **absenteeism** – measured in terms of hours absent from remunerated work (Burton et al., 2017; Hafner et al., 2017; Daley et al., 2009; Ricci et al., 2007; Kessler et al., 2011; Godet-Cayré et al., 2006; & Metlaine et al., 2004)
- **loss of productivity at work** – measured in terms of additional time taken to complete a given task correctly as a result of loss of concentration, repeat work, slower working, or doing nothing at work

32 https://www.stats.govt.nz/topics/gross-domestic-product

(Burton et al., 2017; Hafner et al., 2017; Magnavita & Garbarino, 2017; Daley et al., 2009; Ricci et al., 2007; Kessler et al., 2011; & Metlaine et al., 2004)

- **cognitive ability and self-control** – the self-regulatory resources of an individual largely allow a worker to abstain from workplace deviant behaviour, such as 'cyber loafing', and remain disciplined at work, behave conscientiously and work diligently (Wagner et al., 2012; Christian et al., 2011). Therefore, a decrease in an individual's self-regulatory resources can lead to lower productivity.

The definitions of productivity included in the studies reflect changes in 'input' approach. The literature review suggests that valuing this change in inputs is typically undertaken using the human capital approach; that is, based on the estimated earnings foregone.

However, as Deloitte Access Economics (2011) notes, a loss in productivity of a person will only equate to a loss in productivity to the economy under fairly strict conditions. These are:

- the economy is at full employment so any reduction in hours worked due to sleep health and attributed shares of other illnesses and injuries, or any permanent reduction in labour force participation through early retirement or death, cannot be replaced by employing or increasing hours of other workers
- the income of an individual is proportional to the total value added to production.

The first condition will fluctuate over time as the economy moves into, and out of, full employment. A reduction in labour when labour is scarce will have a greater impact on productivity compared to an economy with an abundant labour supply. In this situation, a temporary or permanent reduction in working hours due to sleep disorders and associated conditions cannot be replaced by hiring another worker. Consequently, a loss in productivity due to sleep disorders and associated conditions is expected to represent a real cost to an economy operating at a low level of unemployment.

The second condition will occur if there is a perfect labour market such that the marginal benefit from an additional hour of work (the value added) is equal to the marginal cost (the wage). In reality, labour markets are imperfect for a number of reasons, for example asymmetric information in the market, and labour market restrictions imposed by government regulation and natural barriers. In addition, synergy created between labour, capital, and land means a reduction in working hours may also impact the productivity of other factors of production.

Consequently, the value of productivity from labour could be greater than or less than the wage provided to an individual, so using lost income as a proxy for lost productivity will tend to either under- or overestimate the true cost. It is likely that in the absence of their condition, people with sleep disorders and associated conditions would participate in the labour force and obtain employment at the same rate and average weekly earnings as others. The implicit assumption is that the numbers of such people would not be of sufficient magnitude to substantially influence the overall clearing of labour markets, and average wages remain the same.

While no studies were found that looked specifically at the effects of noise-related sleep disturbance on productivity, several studies have investigated the economic costs of fatigue, sleep disorders, and poor sleep, where the costs include any or all of:

- the **direct health costs**, including expenditure on hospital care, health practitioners, pharmaceuticals, diagnostic tests, health aids and appliances, research, community and public health, and capital and administration (Hillman et al., 2018; Hillman et al., 2006; Deloitte Access Economics, 2011; Daley et al., 2009). These costs are typically obtained from national health databases and then attributable fractions are applied to control for the various conditions often associated with sleep disorders
- the financial costs of **work-related injuries**, excluding disease and health costs (Hillman et al., 2006; Deloitte Access Economics, 2011; Daley et al., 2009)

- the financial costs of **motor vehicle accidents**, excluding health costs (Hillman et al., 2006; Deloitte Access Economics, 2011; Daley et al., 2009)
- the lost earnings associated with **absenteeism and lower productivity** (Hillman et al., 2018; Hafner et al., 2017; Ricci et al., 2007; Kessler et al., 2011; Daley et al., 2009; Deloitte Access Economics, 2011; Godet-Cayré et al., 2006; & Metlaine et al., 2005)
- the **burden of disease**, measured in DALYs (Hillman et al., 2006; Deloitte Access Economics, 2011; WHO, 2008; & Rhodes et al., 2013).

The studies reviewed typically examine four broad types of potential productivity losses, which are:

- **premature workforce separation** – early retirement or other reasons for workforce withdrawal
- **temporary absenteeism** – due to being unwell more often than average and taking time off work, while remaining in the workforce
- **lower productivity at work** (‘presenteeism’) – producing less due to reduced hours or lower capacity while at work
- **premature mortality** – the discounted net present value of the future income streams that would have been earned if a person dies prematurely.

It is worth noting here that several of the studies that have examined the economic costs associated with fatigue and sleep disorders (eg, Daley et al., 2009; Ricci et al., 2007; Kessler et al., 2011) arrive at similar estimates of the magnitude of losses incurred for both present and absent workers, suggesting that these estimates may be used interchangeably.

It has also been shown that sleep loss is associated with significantly higher levels of productivity impairment amongst employees who also have one or more chronic health conditions such as hypertension, asthma, diabetes, and congestive heart failure. For example:

- A comprehensive review by Ozminkowski et al. (2007) of research into the effects on productivity of employees with comorbid insomnia, revealed that the majority surveyed reported that **insomnia often occurs in the presence of other medical and psychiatric conditions**, such as depression, anxiety, restless leg syndrome, or painful illnesses. Other correlates of insomnia included fatigue, reduced physical ability, impaired social performance, and higher rates of absenteeism from work, accidents at work, and presenteeism (ie, lower productivity while at the workstation).
- A number of other international studies of insomnia and work performance have been carried out that have consistently found **insomnia to be related to either short-term absenteeism** (Gureje et al., 2007; Leger et al., 2006; Westerlund et al., 2008, all cited in Kessler et al., 2011; Daley et al., 2009), **presenteeism** (Daley et al., 2009; Godet-Cayré et al., 2006) **or disability** (Sivertsen et al., 2006; Jones et al., 2013), but did not monetise results. A number of these previous studies examined the relative importance of insomnia in predicting presenteeism and absenteeism (Daley et al., 2009; Bolge et al., 2009; and Godet-Cayré et al., 2006). They uniformly found that insomnia is much more strongly related to presenteeism than absenteeism. This means that workers with insomnia generally put in the same number of work hours as other workers, but that their on-the-job performance is lower than other workers. This finding is consistent with a larger literature review showing that the majority of lost work performance occurs during days when workers are on the job rather than off work (Kessler et al., 2011).
- Ricci et al. (2007) found that when fatigue co-occurs with other conditions, it is associated with a threefold increase, on average, in the proportion of workers reporting lost productive time. Similarly, Kessler et al. (2011) and Hillman et al. (2006) found that the number of days of lost work performance for workers due to presenteeism with insomnia was significantly less than those with co-morbid conditions.
- **Psychosocial factors can affect the prevalence of insomnia in workers**, which in turn can affect productivity. These include perceived job stress, social support; interpersonal conflict with close co-

workers; perceived work environment; jobs with high demands and low job control; job dissatisfaction; and poor job performance (Metlaine et al., 2004).

As noted above, there are several studies that examine the nature and significance of economic costs – including productivity losses – associated with sleep disorders and poor sleep. For example:

- On the basis of telephone interviews with a national sample of 7,428 employed health plan subscribers, Kessler et al. (2011) examined the effects of insomnia on work performance (net of comorbid conditions), in terms of both absenteeism and presenteeism. The researchers found a significant relationship between insomnia and low on-the-job performance (presenteeism) but not absenteeism. This means that workers with insomnia generally put in the same number of work hours as other workers, but that their on-the-job performance is lower than other workers. For each worker diagnosed with insomnia, absenteeism was found to have a mean value of 7.1% (equivalent to less than one and a half days of absence in a 20-day work month) while comparable values for presenteeism are a mean of 14.2% or 2.84 days per 20-day work month. This suggests that most of the lost work performance occurs during days when workers are on the job rather than absent. The annualised association of insomnia with presenteeism, controlling for socio-demographics was 11.3 days of lost work performance for each worker with insomnia before controlling for comorbidity. This estimate reduced to 7.8 days when controls were introduced for comorbid conditions.
- Ricci et al. (2007) conducted telephone interviews with a sample of 11,719 workers who had screened positive for fatigue in the previous two weeks in order to estimate the prevalence of fatigue and cost of fatigue-related lost productive time (LPT) in the US workforce. LPT was measured as the sum of self-reported hours per week absent from work for a health-related reason (ie, absenteeism) and the hour-equivalent per week of self-reported health-related reduced performance while at work (ie, presenteeism). Presenteeism was quantified by measuring the average frequency of engaging in five specific work behaviours and the average amount of time between arriving at work and starting to work on days not feeling well. Lost labour costs were estimated by converting hours of LPT into lost dollars using self-reported annual salary or wage.
- Daley et al., (2009) randomly selected a sample of 948 adults from the province of Québec, Canada and asked them to complete questionnaires on sleep, health, use of healthcare services and products, accidents, work absences, and reduced productivity with the objective of estimating the economic costs of insomnia. Participants answered a questionnaire designed to obtain information on costs associated with healthcare service and product utilisation, use of alcohol as a sleep aid, hospitalisations, productivity, absenteeism, and accidents.
- Another study (Guertler et al., 2015) on the relationship of lifestyle behaviours with presenteeism in Australia explored participants' ratings of their overall job performance and compared these against a sleep duration and quality question which revealed a significant association between presenteeism and poor sleep quality.
- Studies by Katz et al. and Burton et al. explored the link between healthy sleep behaviours and employee productivity using employee health assessments, and self-reported average hours of sleep and productivity measures respectively, with both studies finding a strong relationship between the hours slept and productivity lost.
- A more recent study by Hafner et al. (2017) across five different Organisation for Economic Co-operation and Development (OECD) countries examined the economic burden of insufficient sleep and found that insufficient sleep is closely related to a wide range of public health issues. The study found that annually within the UK, around 0.2 million days of work time is lost due to insufficient sleep and when included with other sleep-related impacts on the supply of labour, a total of around USD$50 billion is lost to the UK GDP due to poor sleep.

Deloitte Access Economics (2017) examined the economic cost of sleep disorders and estimated the economic cost of all sleep disorders in Australia to be around USD$12.19 billion, building on the work of previous reports by the same authors (Deloitte Access Economics, 2004, 2011). Using estimates of the prevalence of three types of sleep disorders within the Australian population in general, the study estimates (using PAFs) the proportion of other health conditions, motor vehicle accidents, and workplace injuries attributable to each sleep disorder and the financial and non-financial costs associated with each of these. The financial costs include direct expenditure on health care and the indirect costs associated with:

- lost productivity due to premature workforce separation and mortality, and absenteeism
- the deadweight loss of raising revenue to fund lost productivity, public health expenditure, social security payments, and a number of costs associated with motor vehicle accidents that were due to sleep disorders
- informal care and other costs of motor vehicle and workplace accidents
- the lower quality of life experienced by someone with a sleeping disorder, estimated on the basis of DALYs.

The ways in which these were measured are summarised in Table A.7 below. Note, however, that although the authors recognise that sleep disorders are significantly associated with lost work performance due to presenteeism, the costs of presenteeism were not included in the study because of the difficulties in obtaining Australian data on the difference in productivity between people with and without a particular illness or injury.

Table A.7 Approaches to measuring components of lost productivity (compiled using information from Deloitte Access Economics, 2017)

Measure	Measurement Approach
Premature workforce separation	The number of people unemployed as a result of a sleep disorder or a condition that is caused by a sleep disorder. To avoid the risk of double-counting, the cost was based on the share of the cost of injuries and illness attributable to sleep disorders. It was assumed that a sleep disorder does not compound the probability of premature workforce separation for people with a particular illness or injury and a sleep disorder compared with those who have that illness or injury but not a sleep disorder. The cost to productivity for each injury and illness associated with a sleep disorder was determined by the difference in the employment rate of people with each injury or illness derived from a sleep disorder and the age gender standardised Australian population equivalent. The loss of employment was calculated separately for each condition using information on the impact on employment rates for the relevant condition. All employment data for Australia was sourced from the Australian Bureau of Statistics.
Premature mortality	The number of premature deaths due to each of the illness or injuries attributed to a sleep disorder based on national Cause of Death data. The productivity lost due to premature deaths was calculated by multiplying the estimated number of deaths from each condition that can be attributed to obstructive sleep apnoea by lifetime potential earnings at the time of death. Lifetime earnings were based on age and gender and adjusted for the probability of employment, full-time or part-time. Assuming a retirement age of 65, the remaining years of employment were calculated for each age group based on the average for a person of each age in the group. The annual productivity loss from premature death was valued using 2010 average annual earnings data by workforce age group. Future streams of income were discounted to a present value using an appropriate discount rate.
Absenteeism	Absenteeism was measured by looking at the number of workdays missed by people with chronic conditions relative to the rest of the population.

	Note that the cost of temporary absence due to the event of an accident was included in the estimates of premature workforce separation. However, people injured in a motor vehicle or workplace accident could also incur an absenteeism cost once they return to work because they may have ongoing health issues as a result of their injury, which require days off work. Due to data limitations this cost could not be calculated.

The Deloitte Access Economics (2011) study measured absenteeism by looking at the number of workdays missed by people with chronic conditions relative to the rest of the population. A survey by the Australian Institute of Health and Welfare (AIHW) (2009) found that people with chronic disease, including depression and cardiovascular disease, reported missing 0.48 days of work per fortnight compared with 0.25 days per fortnight missed by people without a chronic disease. This amounts to an average of 11.5 days of sick leave per year for a person with a chronic condition compared to six days for a person without a chronic disease. The cost of absenteeism for each condition caused by a sleep disorder was therefore assumed to be 5.5 days per person employed. Note that this figure relates to absenteeism from chronic disease but, on the basis that sleep disorders contribute to chronic disease, is used by the Deloitte Access Economics (2011) study as a measure for absenteeism specifically from sleep-related chronic disease.

All the other studies reviewed that estimated the productivity losses associated with both absenteeism and presenteeism found that the majority of the costs attributed to lost productive time are due to reduced performance while at work, not work absence (Ricci et al., 2007; Kessler et al., 2011). Workplace losses are typically linked to decreased output and lower productivity levels resulting from a loss of cognitive ability and self-control.

The cost estimates (expressed in terms of days / hours of lost productive time) published in these studies are considered suitable for transfer to the New Zealand context (with appropriate adjustments) given the similarities in socio-economic conditions and workforce structures.

Key issues

There are several methodological difficulties in deriving estimates of the productivity losses associated with noise-related sleep disturbance. Key amongst these will be isolating the effects of noise-related sleep disturbance in the first instance, and then the effects of this on productivity. Muirhead et al. (2011) identify some of the issues in measuring sleep-related productivity losses:

- Much of the evidence linking noise to sleep disturbance is based around subjective self-reports which may not correlate with other objective measures of sleep quality.
- While studies can identify distinct control groups, when considering the whole population, it may be difficult to discriminate between noise-induced sleep disturbance and disruption caused from other factors such as lifestyle, physiology, and lighting.
- It may be possible to make estimates based on L_{night} noise values since some studies show a correlation, however noise events have been shown to be a more accurate measure of awakenings.
- Measuring productivity in terms of efficiency in the workplace is difficult to do objectively and, as such, most studies rely on either subjective assessments or accident rates and absence only.
- Estimates of the economic costs of sleep disturbance vary in terms of the factors affecting productivity that were included and the population of the country of interest. The necessary precautions would therefore need to be taken in using these for the purposes of value transfer for the present study.

In its 2014 guidance, DEFRA sets out an interim approach for valuing the productivity losses associated with noise-related sleep disturbance. This involves taking average low and high estimates of productivity losses from a range of existing studies covering the economic costs of insomnia and insomnia syndrome (Kessler et al., 2011; Godet-Cayre et al., 2006; Rosekind et al., 2010; Daley et al., 2009), and sleep disorders

(Uchiyama, 2013; Deloitte Access Economics, 2011), adjusting them for purchasing power parity and differences in price year. These are then combined with the recommended exposure-response function for determining the percentage of the population that is highly sleep disturbed and assuming that one employee per dwelling is affected. The percentage highly-sleep disturbed (HSD) population is then multiplied by the number of affected dwellings and the estimated average productivity loss per year. It is noted that while the estimates use average estimates from five studies to take account of variation and bias, the following uncertainties, to some extent, are still likely to remain:

- A national average productivity loss per employee is difficult to estimate as it is difficult to compare time loss with output between different jobs.
- The above issue is further compounded by taking average productivity loss per employee from different country studies, as relative incomes are different – although this has been mitigated to some extent by scaling by the ratio of GDP per capita estimates, there is still a risk that not all income variation is accounted for between countries.
- There may be other factors in or around the office that affect average productivity through higher stress and lower concentration levels that may not be reflected in the above estimate.
- Given that only five studies have been used to provide an average estimate for productivity loss per employee, this may not fully take account of all biases to do with a particular study, for example does the sample population reflect the true population?
- The productivity cost estimates only look at financial productivity loss to a company and not the economy, so spill-overs from investment and tax from this productivity loss are not factored in, the figures could therefore be an underestimate of the productivity loss to the economy.

Finally, the HAPINZ 2.0 study (Emission Impossible et al., 2012) included the costs of lost income as a measure of the loss of economic output. This assumes, that in a competitive economy, the gross income paid to a worker represents the value of the output produced. Lost economic output was estimated on the basis of the average daily income across the whole population[33]. Lost income was associated with the loss of output during hospitalisation and restricted activity days (RADs).

Given the difficulties in estimating the costs of productivity losses while at work, the approach in the SCON study is likely to consider the costs of absenteeism only. Following the lead of the HAPINZ 2.0 study, lost productivity will be measured in terms of the lost income during hospitalisation and as a result of restricted activity. Given the relatively short duration of absenteeism per case, it is assumed that employers do not incur costs for replacement staff. It is further assumed that any sick pay during absence is equivalent to the average daily wage rate. The loss of economic output is therefore estimated on the basis of either the lost employee income or the costs of sick pay (which are equivalent). These are not additive for the reasons stated above; to combine them would result in double-counting.

The relationship between academic performance and lifetime earnings

The relationship between academic performance and salary, lifetime earnings, or occupational performance has been examined. Oreopoulos (2007) states that teenagers who do not 'drop out' of compulsory schooling are likely to increase their lifetime earnings by 15% for one additional year of education, and are less likely to be unemployed, thereby having a positive impact upon GDP. Tyler (2004) investigated the relationship between the performance of 16 to 18-year-old dropouts and earnings by means of a maths test, and it was

[33] This is calculated as the average weekly pre-tax income (from wages, salaries and self-employment) for the working age population (those 15 years and over, including those working and not working) divided by the total estimated resident population; this is then divided by seven to obtain an average daily income.

found that those who achieved results one standard deviation above the mean enjoyed an average increase in salary of approximately 6.5% in their first three years in employment.

A report for the UK Commission for Employment and Skills (Garrett et al., 2010) collated information on the value of skills, reviewing a number of independent studies of skill value at national level as part of the report. The report cites a number of papers which demonstrate the benefits of an improved education: Bassanini and Scarpetta (2001) found, in a study of 21 OECD countries from 1921 to 1998, that on average, one additional year in education is associated with a long-run increase in output of 6%. Sianesi and Van Reenen (2003) found that such an increase in education has an impact of 3–6% on the level of output and of over 1% on the growth rate.

There have been a significant number of studies undertaken looking at the impacts of noise on academic performance and the impacts of academic performance on GDP. However, no studies have been identified where a direct link between the three factors has been examined or established.

In terms of the mechanism linking the effects of noise exposure in an academic environment to productivity in the workplace, no studies have been identified which address the complete mechanism.

Estimating the value of productivity losses associated with other health outcomes arising from noise disturbance

The 2012 HAPINZ study (Emission Impossible et al., 2012) considered productivity losses as a result of hospitalisation (absenteeism) and RADs. Loss of output during hospitalisation was calculated using information on average weekly wage rates and the average length (in days) of a hospital stay for each of cardiovascular and respiratory conditions for adults and children. Estimates of effects on hospital admissions for respiratory diseases in children were based on the results of a multi-city Australasian study (Barnett et al., 2005) and the effect on hospital admissions in adults, based on the results of a European meta-analysis, APHEIS (2004), as cited in a European guide to air pollution impact assessment (ENHIS, 2007).

The cost of RADs was based on estimates of the number of RADs from an American study and the average loss of output per day, again based on average weekly wage rates.

It is proposed that the SCON model adopts a similar approach to that being developed in the current refresh of the HAPINZ study.

A3.10 Approaches to valuing the impacts of transport noise exposure on amenity

There are two generally accepted ways to value amenity[34]. One is to estimate the ‘burden of annoyance’ by combining exposure data with noise annoyance exposure-response relationships to first obtain a measure of the number of households or individuals that are highly annoyed. The number of highly annoyed people is then translated into a health impact using a recommended disability weight to calculate DALYs which are then monetised using a monetary value of a QALY or a VoLY. This is the approach suggested by the WHO (WHO, 2011, 2018) and others (for example, EEA, 2020; Defra, 2014) and recognises noise as a health outcome which is based on physiological effects which can be measured objectively across individuals.

A major limitation of this approach relates to the difficulties in weighting annoyance and relating it to existing weighted outcomes, particularly given the wide range in estimates of the disability weight (Berry & Sanchez, 2014). Another significant issue is whether or not annoyance significantly contributes to disability, and it should be considered in the noise-induced burden of disease. This is discussed further below.

[34] Note that the terms amenity and annoyance are used interchangeably in the literature.

The other recognises that an individual's response to noise depends on the individual's attitude and sensitivity to noise and relies on the traditional estimation of the WTP to avoid or to accept a certain level of noise. This assumes that annoyance is the driver behind individuals' willingness to pay or accept a certain level of noise and it does not require the link to be quantified. The economic valuation can be undertaken using either revealed preference (eg, hedonic pricing) or stated preference (eg, contingent valuation) techniques.

Hedonic pricing uses house market prices as a proxy of the preference that consumers revealed for noise. Numerous studies have been conducted to ascertain the effects of transport noise on property values (Tinch, 1995; Bateman et al., 2001; Navrud, 2002; Nelson, 2004; Day et al., 2006). These have all reported a negative effect (ie, each decibel increase in noise level causes a reduction in the price of the affected house).

In contrast, few studies have specifically examined the effects on property prices of a reduction in noise. Policy studies and transport appraisal guidance (Nellthorp et al., 2007; Defra, 2014; Department of Transport, 2020) treats a 1 dB increase in noise from a given level as equivalent to a 1 dB decrease in noise to that level, recognising that marginal WTP increase with the level of noise experienced. Although amenity values derived from hedonic pricing studies are no longer included in transport appraisal in the UK, in the past, these were simply provided as damage or benefit costs (depending on whether an increase or decrease in noise was experienced) for each 1 dB change in noise level (by source) for a given year and then adjusted to account for the effects of inflation and income growth.

Stated preference uses questionnaires in which people state their preferences based on hypothetical situations. There is a significant amount of research already available and established methodological techniques, especially for hedonic prices (see for example, Bateman et al., 2001; Navrud, 2002; Nelson, 2004; Wadud, 2013; Bristow & Wardman, 2015).

These approaches are described in more detail in Table A.8.

Table A.8 Approaches to valuing noise nuisance

Approaches to valuing noise nuisance (from Bristow & Wardman, 2015)
Revealed preference approaches Noise nuisance has commonly been valued using hedonic pricing (HP), a revealed preference approach which uses the market for a particular good, in this case the housing market, to estimate the value of the different components of the good. The HP method is attractive because it has a basis in real decisions in the marketplace and, at least until recently, underpinned many values used in transport appraisals in Europe (Bristow et al., 2015). Earlier versions of the UK Department for Transport's Transport Appraisal Guidance (webTAG), for example, used hedonic pricing approaches to value the noise impacts of transport schemes. The value of noise obtained is usually expressed in the form of a Noise Depreciation Index (NDI) or Noise Sensitivity Depreciation Index (NSDI) which indicates the percentage change in house prices that results from a 1 dB change in noise levels. The number of HP studies on aircraft noise is such that a number of meta-analyses have been carried out. Wadud (2013) identified 65 NDI values ranging from 0% to 2.3% and included 53 estimates in a meta-analysis concluding that a 1 dB increase in aircraft noise levels leads to a fall in house prices of between 0.45% and 0.64%. This estimate is broadly consistent with meta-analysis by Nelson (2004) and earlier review by Nelson (1980), though somewhat lower than the estimates of Schipper et al. (1998) of 0.9% to 1.3%. There are fewer studies of road traffic noise and no substantive meta-analysis has been conducted. Bateman et al. (2001) reviewed 18 studies of road traffic noise mostly from North America, finding a range from 0.08% to 2.22% and an average NSDI of 0.55%. Comparison of studies is difficult due to differences in functional form, the quality and scope of data, definitions of variables, and the level of discrimination of the impact being valued.

Although the HP approach is broadly accepted and has traditionally underpinned many values used in public sector appraisals, the range of values is nonetheless large and, moreover, this variation is largely unexplained. Furthermore, the revealed preference approach is based on the assumption that there is perfect labour and personal mobility and that individuals are well-informed about the risks they face in exposure to noise. The difficulty in fulfilling these requirements is thought to explain the variation in estimates produced by revealed preference studies (Dolan & Metcalfe, 2008). HP is also limited in that it can only give a value of disturbance as experienced at home and many studies that have been undertaken only control for differences in the characteristics of properties (size, etc) and fail to control for other factors that may influence house prices and consumer behaviour (eg, air quality, local environmental quality). Meta-analysis suggests that this cost may be capitalised through a house price discount of about 0.5% to 0.6% per dB(A). However, this cannot inform what people might be willing to pay now for changes in the noise level experienced or how this might vary by time of day, day of week, or season (Bristow & Wardman, 2015).

Stated preference approaches

Given the difficulties posed to the revealed preference approach by imperfect markets and a lack of data, economists have turned to stated preference approaches to value non-market goods. Within the class of stated preference methods, there are two alternative groups of techniques: choice modelling (CM) and contingent valuation (CV). In general, contingent valuation concentrates on the non-market good or service as a whole (eg, WTP for a defined change in noise levels), while choice modelling seeks people's preferences for the individual characteristics or attributes of these goods and services (eg, preferences for aircraft versus road noise or different levels or durations of noise, etc). The advantage of contingent valuation questions is the ability to elicit exactly the information that is required (Nellthorp et al., 2007).

The main challenge is the necessary assumption that individuals have a coherent set of preferences. A number of phenomena have been identified as evidence that such coherent preference may not be observed in practice, including: substitution effects; endowment effects; hypothetical bias; the influence of irrelevant cues, where respondents are influenced by the elicitation procedure, such as start-point bias, anchoring effects, focusing effects, embedding effects, and range bias (Dolan & Metcalfe, 2008). CM techniques have been developed largely to take account of some of the shortcomings of CV and have been increasingly applied in this context. Another major issue with stated preference approaches, specifically as applied to noise, is that people may not have a good understanding of what a given change in noise level (eg, 1 dB, 5 dB) means although this can be overcome through techniques such as auralisation and simulation of soundscape scenarios.

As might be expected, the SP valuations of noise nuisance exhibit a wide range. This variation may be explained by variations in data type and survey method, the systematic influence of study and country specific factors and, importantly, intertemporal effects.

The WHO Environmental Noise Guidelines (WHO, 2018) consider long-term annoyance and sleep disturbance due to noise to be important health outcomes. According to the WHO definition of health, which is "a state of complete physical, mental and social wellbeing and not merely the absence of disease or infirmity" (WHO, 2006, p.1), documenting only physical health does not present a complete picture of general health. The importance of considering both annoyance and self-reported sleep disturbance as health outcomes is further supported by evidence indicating that they may play a part in the causal pathway of noise-induced cardiovascular and metabolic diseases (Eriksson et al., 2018; WHO, 2018). Following a systematic review of the noise-health literature, van Kamp et al. (2018) found that the most prominent argument in the literature against the inclusion of annoyance as a health endpoint in DALY calculations, is that it does not have an International Classification of Diseases (ICD) code.

Although it may be argued that the inclusion of annoyance as a health impact may introduce a risk of double-counting if, in health terms, it is simply a precursor to other health impacts, others (DEFRA, 2014; Bristow & Wardman, 2015; WHO, 2018) have argued that annoyance from noise clearly impacts on wellbeing and thus its inclusion is wholly compatible with the WHO 1946 definition of health (WHO, 2006).

This then raises concerns that valuing noise using both WTP or WTA approaches and in terms of health impacts would result in double-counting (DEFRA, 2014). To address this, DEFRA (2014) recommends the use of DALYs to reflect the value of impact on public annoyance, noting that this approach has a range of additional benefits, including:

- using a consistent approach with other impacts reduces the risk of double-counting. For example, it reduces overlap with sleep disturbance and wider potential explanatory variables
- focusing on an exposure-response function clarifies the pathways through which any values are derived and can help identify key research areas.

Finally, as relationships are available by transport mode it is possible to provide bespoke estimates for road, rail, and aviation noise.

The other major advantage of this approach is that it allows separate factors for road, rail, and aviation noise for annoyance, better reflecting the existing evidence base (DEFRA, 2014).

Furthermore, as noted by Day et al*.* (2007), while there are numerous studies from which values derived via means of revealed and stated preference approaches could be transferred, doing so is difficult in theory and practice, not least of all because of differences in income across countries, but also because they may not reflect current preferences of residents, especially for older studies.

Bristow and Wardman (2015) argue against use of the health impact approach on the basis that values derived from hedonic pricing and stated preference studies are less likely to include the more serious health effects (cardiovascular diseases and metabolic effects). This is because the relationships between noise and health are not widely understood, partly because the evidence base is still developing. They therefore consider it unlikely that combining values from HP and SP studies for amenity effects with those from an impact pathway approach for more serious health effects would lead to double-counting. Moreover, on the basis of a meta-analysis, they found that in practice, in the UK context, the impact pathway approach (using DALYs) produced lower values whereas it might be expected that it would yield higher values if it includes the more serious health effects that are unlikely to be factored into WTP estimates. They attribute this, at least in part, to the effective zero value attributed to all who are not highly annoyed or highly sleep disturbed.

A.3.11 Conclusions and recommendations

While the literature relating to the economic value of transportation noise is relatively small, there is a significant volume of literature published on health economics that details the conceptual approaches to quantifying and valuing health impacts and their limitations. Following a review of the relevant literature, it is possible to start formulating a number of recommendations for the social cost of noise model. It is, however, important to note that some of these may need to be revisited in light of both the outcomes of the health assessment (ongoing at the time of writing), particularly with respect to the specific health endpoints to be included in the cost model, and the final methodology for the HAPINZ 3.0 study, with which the social cost of noise model should align as far as possible.

Components of the social cost of noise

The literature identifies a number of components of the social cost of transport noise, including the:

- wellbeing cost to individuals in terms of premature mortality and morbidity
- financial costs associated with hospitalisation, medication, and additional home or childcare that may be needed as a result of illness or disability. 'Avertive' expenditures (eg, installation of double glazing) have also been identified as a potential financial cost but these are not examined in detail in the literature. These are not considered further for the cost model given that the exposure-response functions for health effects are typically based on noise levels measured at the external façade of buildings rather

than indoors; including avertive expenditures would therefore result in some double-counting with health effects
- productivity losses associated with premature mortality and morbidity, including the lost productive time as a result of hospitalisation and reduced capacity to perform while at work
- effects on property prices as a result of annoyance and a loss of amenity
- impacts on the functioning of ecosystems and wildlife. Given the limited research in this area, impacts on ecosystem functioning are not included in the cost model.

Valuing health impacts on premature mortality and morbidity

The specific health outcomes to be valued in the model will be determined on the basis of the findings from the review of health impacts. It is proposed to use a combination of DALYs, QALYs, and VoLYs to quantify and value impacts on mortality in a way that is consistent with the overall framework being adopted in the forthcoming HAPINZ 3.0 study.

Financial costs of illness

While several of the studies identify the direct (financial) costs of illness as a component of the total social cost, few of these go on to include estimates of these costs. This may be because previous work has shown that these tend to be much smaller in overall monetary terms when compared to other health effects. While the specific health endpoints to be included in the model have yet to be identified, the literature review findings suggest that there is sufficient evidence to include the following costs:

- GP visits and medical prescriptions for health effects not requiring hospitalisation (eg, the effects of poor sleep)
- average medical cost per hospitalisation for health endpoints relating to cardiovascular diseases, including emergency or pre-hospital (fixed) costs, the 'hotel' costs, and the follow-on costs after discharge from hospital including any long-term disability.

Productivity losses

The productivity losses associated with noise, such as those caused by sleep disturbance, health effects, workplace distraction, and (in early life) diminished academic performance are not well researched in terms of monetisation. There are substantial gaps in the evidence base and no agreed methodologies in place, particularly with regards to the effects of transport-related sleep disturbance on next-day productivity at work.

Given the difficulties in establishing the value of productivity losses with presenteeism, efforts to include estimates of lost output are typically approximated by the costs of lost income as a result of absenteeism only. This assumes that, in a competitive economy, the gross income paid to a worker represents the value of the output produced. This was the approach adopted in the HAPINZ 2.0 study and is understood to be the approach that is most likely to be used in HAPINZ 3.0 where lost income was associated with the loss of output during hospitalisation and RADs. Given the relatively short duration of absenteeism per case, it is assumed that there are no additional costs to the employer (eg, in relation to recruitment of temporary staff or sick pay).

For consistency, it is proposed to adopt a similar approach to that used in the HAPINZ 3.0 study.

Annoyance and amenity

There is some debate in the literature as to how annoyance should be treated. While traditionally it had been valued on the basis of house price differentials (using hedonic pricing models), more recent studies and guidelines recommend its inclusion as a health impact to avoid the risk of double-counting and in recognition that annoyance is often a precursor to more serious disease. For the purposes of the cost model, it is recommended that annoyance is quantified as a health impact (ie, using DALYs) not only to avoid double-counting with other noise-health impacts but also because few studies (from which estimates of house price differentials could be transferred) control for other confounding factors (eg, air quality).

Appendix B: Social cost (health) of noise dashboard user guide

B.1 Introduction

The social cost (health) of noise for New Zealand's road and rail network has been calculated through noise modelling and economic analysis. An interactive dashboard has been developed to enable the interrogation of the economic model outputs and noise modelling results.

B.2 SCON dashboard

The dashboard is accessible online through the NZTA ArcGIS Online page and enables the user to interact with the spatial data through dropdown bar selections. The dashboard elements (charts, map, and tables) update based on the chosen selections. It is also possible to interact with the map and some of the charts to filter data and zoom to areas. On first display, the dashboard will display a central (base) case of health cost impacts, by territorial authority for all New Zealand.

Figure B.1 SCON dashboard initial view

B.3 Side panel selections

Displayed on the left-hand side of the dashboard, the user is presented with three unique selection criteria: Cost Model Scenario (select one), Area Type (select one), and Area Name (select one or multiple).

Table B.1 Side panel selections

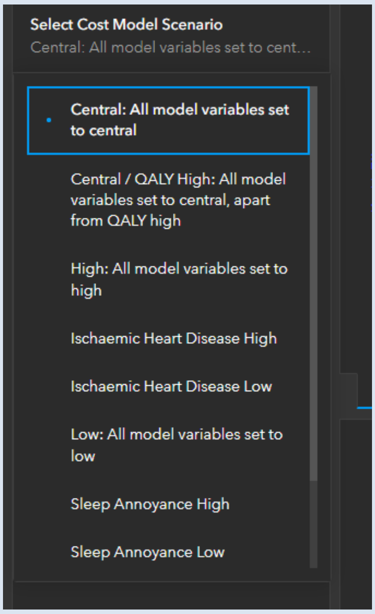
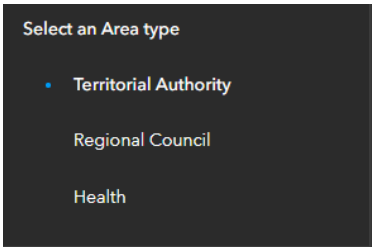
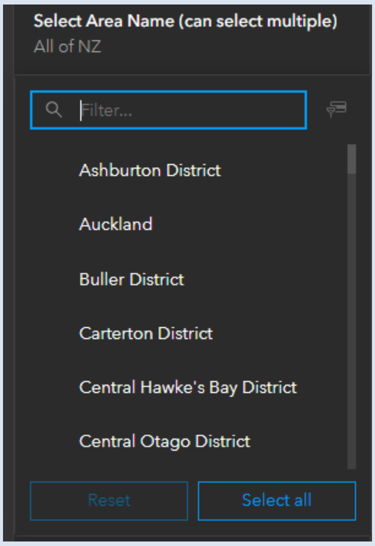
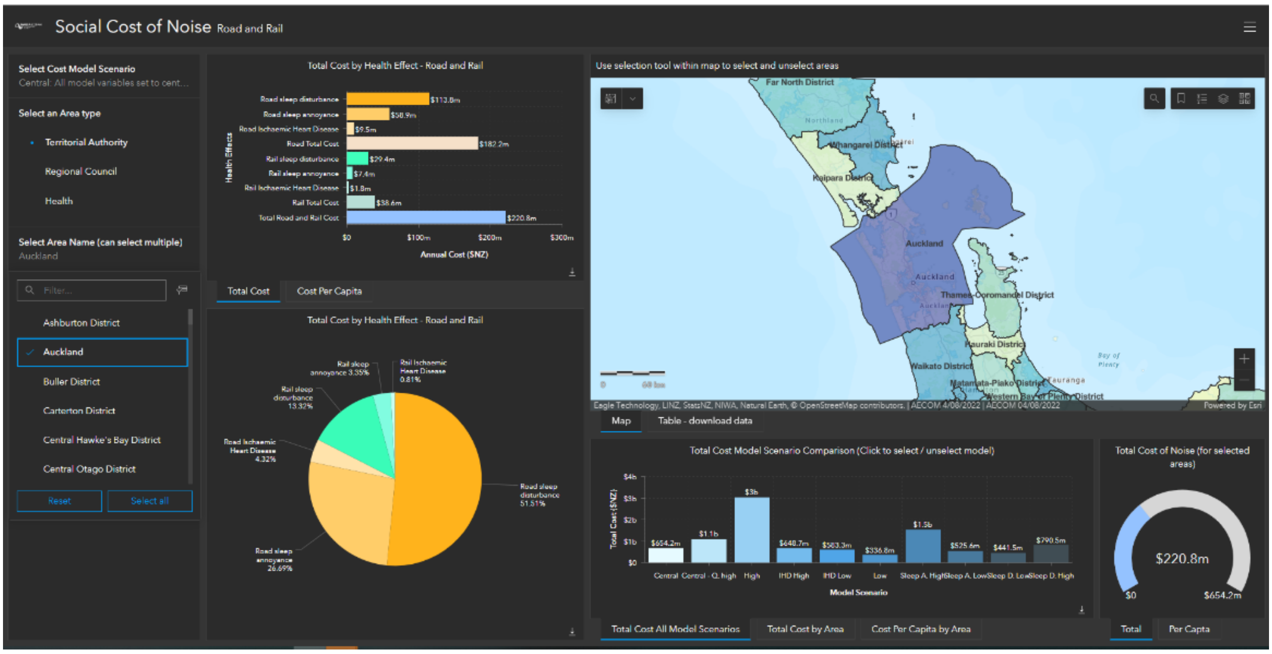
Icon	Description
	The economic cost model scenarios can be selected. There are 10 different scenarios to choose from with a range of model variables that update the dashboard elements. The initial dashboard view displays the central case model scenario.
	The economic cost model has been calculated across different area types. Selection of this dropdown will change the area types displayed on the map and the corresponding economic values.
	The selection of an area name will zoom the map to the area, flash the corresponding area, and update the dashboard elements. Multiple areas can be selected as desired. A reset option exists below the dropdown list to remove all selected areas.

Figure B.2 below shows the updated results from the selected search criteria. The user is now able to view all the graphical information presented, relevant for the criteria selected.

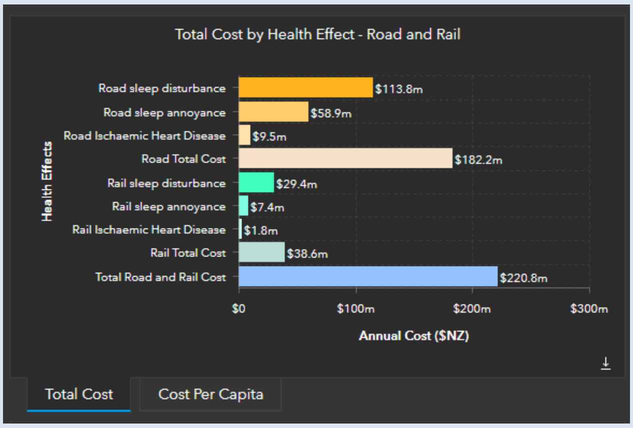
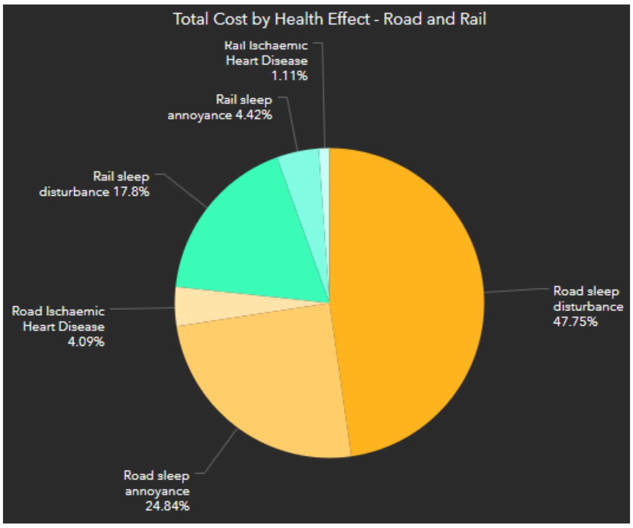
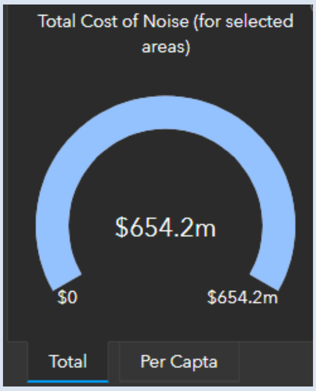
Figure B.2 Example dashboard selection to Auckland



B.4 Charts

Charts are provided throughout the dashboard to present insight to the user. Where data is downloadable from a chart, a small arrow icon is displayed at the bottom right-hand side. Data will be downloaded as a comma separated delimited (.csv) file.

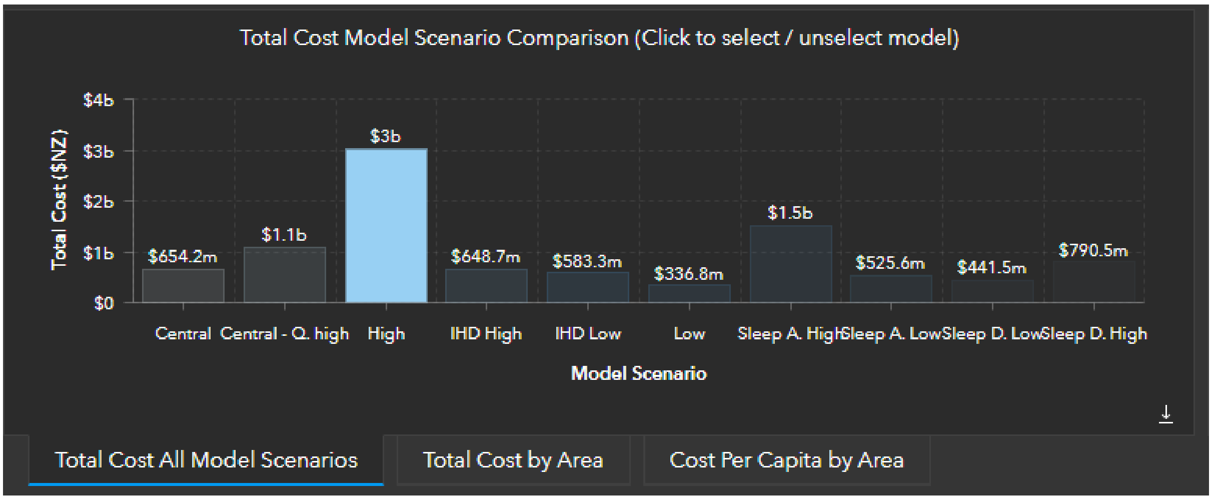
Table B.2 Chart descriptions

Icon	Description
	The bar chart shows each of the three calculated health effects, followed by a total for the respective type of noise. The final bar in series shows the total road and rail cost. Note the functionality at the bottom of the bar chart to toggle the results between Total Cost and Cost Per Capita.
	The pie chart reflects the Total Cost by Health Effect. Results in the chart are expressed as percentages of the total cost. The results are reflected both visually and 'called out' with descriptive text.
	The gauge displays Total Cost or Per Capita Cost, for the selected areas. All areas across New Zealand are shown in the default view. The Total Cost for the selected economic scenario informs the total for the gauge display. As individual areas of New Zealand are selected, the total cost associated with the area will change accordingly

B.5 Selection through charts

Some charts have functionality to inform the selection criteria, which in turn, will update other respective areas of the dashboard.

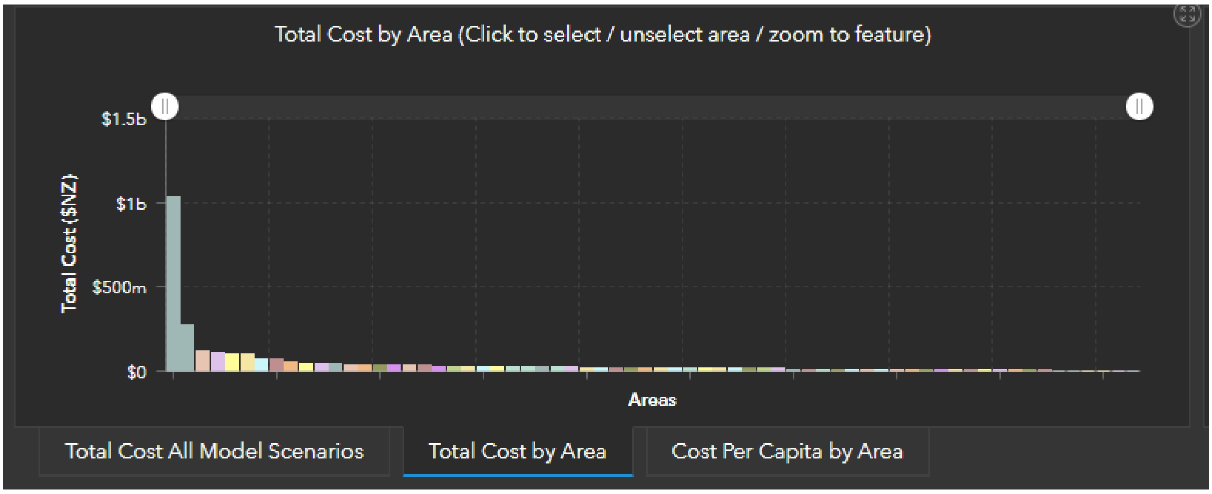
Figure B.3 Cost model scenario



The charts below the map are interactive and filter the data in the dropdowns and dashboard.

Note: the chart must be clicked to select and clicked anywhere on the chart to unselect. Otherwise, the dropdowns will be limited.

Figure B.4 Total Cost by Area or Cost Per Capita charts



The charts will filter the data by area when the bars are selected. The charts Total Cost by Area and Cost Per Capita filter each other. Click last selected chart to unselect and clear filter or select a model scenario or area type dropdown from the sidebar to reset.

The scrollbar at the top will enable zooming into the chart.

B.6 Map controls

Functionality exists within the dashboard map to enhance the user experience and understanding of the data presented.

Figure B.5 Map with legend activated

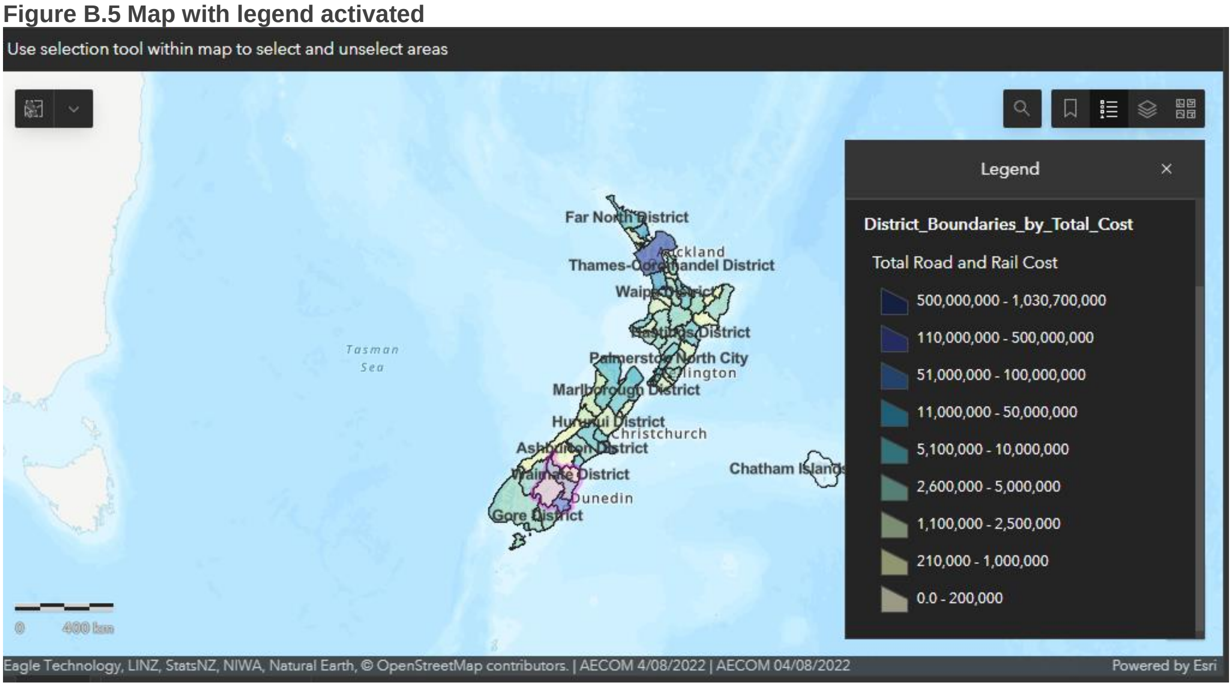
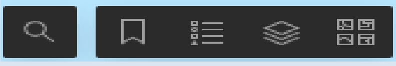
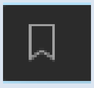
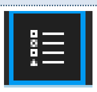
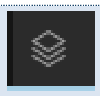
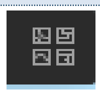
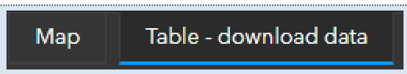


Table B.3 Map controls

Icon	Description
	Map controls, selectable options.
	Address search, enter an address if desired.
	Create a bookmark for a default or preferred view.
	A legend for data presented in the map is displayed upon one click, toggle on or off as desired.
	Layers can be included or excluded from the map by clicking on this icon, then by selecting or deselecting the layer name, eg, ‘District Boundaries’.
	The ‘Basemap’ displayed can be amended to suit the user’s preference or requirement by clicking this icon and selecting any of the options that appear in the dropdown list.
	At the bottom of the map, the user can select ‘Table – download data’, this presented all selected results in a tabulated format. The user can click the small arrow icon at the bottom right corner to download the results. Data will be downloaded as a comma separated delimited (.csv) file.

B.7 Selection using the map

Functionality within the map allows the user to select areas of New Zealand with several unique features.

Figure B.6 Map with legend activated

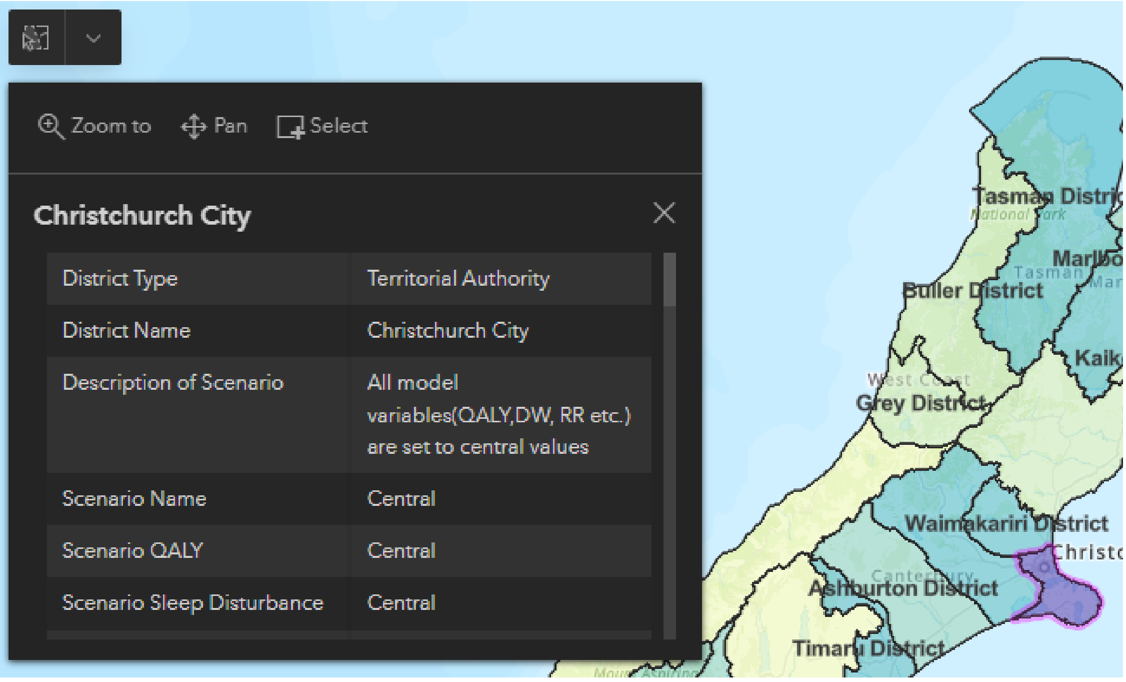
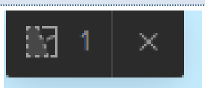
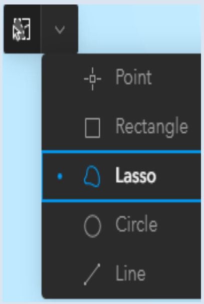


Table B.4 Selection options

Icon	Description
	By selecting a region of the map, a pop-up box will be displayed. The pop-up box will display several data attributes relevant to the region selected. These attributes can be reviewed by utilising the scroll functionality on the right-hand side of the pop-up box. Three options at the top of the pop-up box enable the user to 'Zoom to' the region of map selected, 'Pan' around the map and 'Select'. Utilising the 'Select' function refreshes all other graphical displays in the dashboard with the results of the selected region.
	Where a selection has been made (as noted above), a new box will be displayed above the pop-up. This box informs the user of how many regions have been selected and allows the user to deselect all regions by clicking the cross. In doing so, graphical displays will revert to the information previously displayed.

	By clicking the dropdown arrow in the top-left of the map, a user can utilise one of several options to manually select areas of the map. When selecting either ‘Point’, ‘Rectangle’, ‘Lasso’, ‘Circle’ or ‘Line’, the user can utilise an option by dragging the cursor across the desired section of the map. Graphical results are updated throughout the dashboard, based on selections made and results can be removed by clicking the cross (as noted above).

Appendix C: GRADE reviews

Annoyance

Table C.1 Review of additional studies – annoyance

Publication	Country	Study design	Number of participants	Exposure type	Exposure assessment	Confounders considered	Noise measure for relationships	Reported associations	Risk of bias
Bouzid et al. (2020)	Tunisia	Cross-sectional	1272	Road traffic noise	Measured and modelled	Yes	L_{den}	Association with %HA where L_{den} > 60 dB, OR = 1.097 [1.043-1.154]	1 (Low)
Di et al. (2019)	China	Cross-sectional	1227	Road and rail traffic noise	Measured and modelled	No	L_{den}	Association with %HA observed. For L_{den} > 63 dB %HA higher for rail than road, for L_{den} < 63 dB then %HA higher for road than rail, for road and rail combined the %HA is between the individual values (potential masking effect)	3 (high)
Hong et al. (2018)	Korea	Cross-sectional	1818	Road, rail and aircraft	Measured	No	L_{den}	%HA curves generated for L_{den} 40-80 dB: Aircraft $100/(1+\exp(-0.113 xL_{den} + 6.122))$ Rail: $100/(1+\exp(-0.163xL_{den}+11.12))$ Road: $100/(1+\exp(-0.13xL_{den}+9.993))$	2 (moderate)
Lechner et al. (2019)	Austria	Cross-sectional	1031	Road, rail and aircraft	Modelled	Yes	L_{den}	%HA curves generated and compared to those from WHO (2018) and older EU curves	1 (Low)

Publication	Country	Study design	Number of participants	Exposure type	Exposure assessment	Confounders considered	Noise measure for relationships	Reported associations	Risk of bias
Lefervre et al. (2020)	France (DEBATS study)	Cross-sectional	1244	Aircraft noise	Modelled using INM	Yes	L_{den}	Significant association identified with %HA with OR = 3.04 [2.30-4.02] with exposure-response curves generated which sit between the WHO (2018) and old EU curves	1 (Low)
Schreckenberg et al. (2016)	Germany (NORAH study)	Cohort or longitudinal study	3508	Aircraft noise	Modelled or estimated	Yes	L(day and evening)	%HA relationships established before opening new runway and in the first and second years post opening. The change in %HA was most significant at the lower noise levels, with the %HA curves higher than the former EU curves and more consistent with those from WHO (2018)	2 (moderate)
Sung et al. (2017)	Korea	Cross-sectional	1836	Road traffic noise	Modelled	Yes	L_{den}	Association with %HA observed for population grouped into low and high noise exposures. Level of noise annoyance varies depending on noise sensitivity especially at low noise levels.	1 (Low)
Wothge et al. (2017)	Germany (NORAH study)	Cross-sectional	2962 for aircraft noise, 3006 for road traffic and	Road, rail and aircraft	Modelled	Yes	L_{den} and average sound pressure level	Evaluated relationships for individual sources, but focus on the combined sources which showed a significant association with between annoyance and average sound pressure level for	2 (moderate)

Publication	Country	Study design	Number of participants	Exposure type	Exposure assessment	Confounders considered	Noise measure for relationships	Reported associations	Risk of bias
			2795 for railway					aircraft noise (in combined noise levels), with less significant increase from road/rail. %HA greater where aircraft noise is dominant	

Notes: Older EU exposure-response curve as those published by Miedema and Oudshoorn (1998).

Table C.2 GRADE review of findings – aircraft noise and annoyance (%HA)

Domains	Criterion	Assessment		Grading
		WHO*	Changes from additional studies post WHO**	
Starting point	**Study design: cross-sectional = high quality**	**High quality**	**Unchanged**	**High quality**
1. Study limitations	Quality of majority of studies (risk of bias)	High quality of majority of studies	Moderate to high quality for the majority of studies	No downgrade
2. Inconsistency	Conflicting results	High between study scatter	Unchanged	Downgrade one level
3. Directness	Direct comparison; same populations, exposures, comparators, and outcomes (PECO)	Same PECO	Unchanged	No downgrade
4. Precision	Small sample size or low numbers of events (HA) OR wide confidence intervals	Large study samples	Unchanged	No downgrade
5. Publication bias	Funnel plot indicates	No publication bias	Not reviewed	No downgrade
Overall judgement				Moderate quality
6. Exposure assessment	Statistically significant trend %HA vs L_{den}	Not assessable	Unchanged	No upgrade
7. Magnitude of effect	Fit of logistic regression	Not assessable	Unchanged	No upgrade
8. Confounding adjusted	Effect in spite of confounding working towards the nil	Not assessable	Unchanged	No upgrade
Overall judgement				**Moderate quality**

* WHO evaluation of studies as presented by Guski et al. (2017)

** Additional studies reviewed by DEFRA (van Kamp et al., 2020) and in this assessment

Table C.3 GRADE review of findings – road traffic noise and annoyance (%HA)

Domains	Criterion	Assessment		Grading
		WHO*	Changes from additional studies post WHO**	
Starting point	**Study design: cross-sectional = high quality**	**High quality**	**Unchanged**	**High quality**
1. Study limitations	Quality of majority of studies (risk of bias)	High quality of majority of studies	Moderate to high quality for the majority of studies	No downgrade
2. Inconsistency	Conflicting results	High between study variance	Unchanged	Downgrade one level
3. Directness	Direct comparison; same PECO	Same PECO	Unchanged	No downgrade
4. Precision	Small sample size or low numbers of events (HA) OR wide confidence intervals	Large study samples	Unchanged	No downgrade
5. Publication bias	Funnel plot indicates	Small publication bias	Not reviewed	Downgrade one level
Overall judgement				Low quality
6. Exposure assessment	Statistically significant trend %HA vs L_{den}	Almost all studies show statistically significant exposure-response relations	Statistically significant relationships in most additional studies	Upgrade one level
7. Magnitude of effect	Fit of logistic regression, Weighted mean r>0.5	Weighted mean r=0.325	Unchanged	No upgrade
8. Confounding adjusted	Effect in spite of confounding working towards the nil	No adjustments	Unchanged	No upgrade
Overall judgement				**Moderate quality**

* WHO evaluation of studies as presented by Guski et al. (2017)

** Additional studies reviewed by DEFRA (van Kamp et al., 2020) and in this assessment

Table C.4 GRADE review of findings – rail traffic noise and annoyance (%HA)

Domains	Criterion	Assessment		Grading
		WHO*	Changes from additional studies post WHO**	
Starting point	**Study design: cross-sectional = high quality**	**High quality**	**Unchanged**	**High quality**
1. Study limitations	Quality of majority of studies (risk of bias)	High quality of majority of studies	Moderate to high quality for the majority of studies	No downgrade
2. Inconsistency	Conflicting results	High between study variance	Unchanged	Downgrade one level
3. Directness	Direct comparison; same PECO	Definition of HA differs between studies	Unchanged	Downgrade one level
4. Precision	Small sample size or low numbers of events (HA) OR wide confidence intervals	Large study samples	Unchanged	No downgrade
5. Publication bias	Funnel plot indicates	No indication of publication bias	Not reviewed	No downgrade
Overall judgement				Low quality
6. Exposure assessment	Statistically significant trend %HA vs L_{den}	All studies show statistically significant OR	Unchanged	Upgrade one level
7. Magnitude of effect	Fit of logistic regression	Most studies provide R2>0.1	Unchanged	No upgrade
8. Confounding adjusted	Effect in spite of confounding working towards the nil	No adjustments	Unchanged	No upgrade
Overall judgement				**Moderate quality**

* WHO evaluation of studies as presented by Guski et al. (2017)

** Additional studies reviewed by DEFRA (van Kamp et al., 2020) and in this assessment

Cardiovascular disease

Table C.5 Review of additional studies – cardiovascular disease

Publication	Country	Study design	Number of participants	Exposure type	Exposure assessment (method and range)	Confounders considered	Noise measure for relationships	CV endpoints	Reported associations	Risk of bias
Andersson et al. (2020)	Sweden	Cohort (longitudinal)	6304 (male only)	Road traffic	Measured and modelled, 50–60 dB	Yes including corrections for NOx	$L_{Aeq,24hr}$ (and L_{den})	IHD incidence, stroke incidence, AF, CVD mortality and natural mortality	Non-significant increased risk (with exposure-response relationships provided) of cardiovascular disease indicators. No association with AF.	3 (high)
Shin et al. (2020)	Canada (Toronto)	Cohort (longitudinal)	701174	Road traffic	Modelled (validated with measured data) (56 dB $L_{Aeq,24hr}$ average)	Yes, including corrections for UFP and NO2	$L_{Aeq,24hr}$ and L_{Aeq}, night	Incidence of hypertension	Association between all noise measures and hypertension	1 (low)

Metabolic diseases and obesity

Table C.6 Review of additional studies – metabolic disease and obesity

Publication	Country	Study design	Number of participants	Exposure type	Exposure assessment (method and range)	Confounders considered	Noise measure for relationships	Health endpoints	Reported associations	Risk of bias
Diabetes										
Jorgensen et al. (2019)	Danish Nurse Cohort	Cohort (longitudinal)	28,731 female nurses over 25 years	Road traffic	Measured and modelled, 50–60 dB	Yes including corrections for lifestyle factors and air pollution	L_{den}	Incidence of type 2 diabetes	No association between long-term exposure to road traffic noise and diabetes after adjusting for PM2.5, suggestive evidence of urban areas (positive association with L_{den} 1.26 [0.97-1.62])	2 (moderate)
Pedersen et al. (2017)	Danish	Cohort (longitudinal)	72,745 individuals – pregnancies between 1996 and 2002	Road traffic	Modelled	Yes including corrections for air pollution	L_{den}	Gestational diabetes	No evidence of an association between noise and gestational diabetes.	2 (moderate)
Shin et al. (2020)	Canada (Toronto)	Cohort (longitudinal)	701,174	Road traffic	Modelled (mean noise level of 52.8 dB)	Yes, including corrections for UFP and NO2	$L_{Aeq,24hr}$ and $L_{Aeq,night}$	incidence of diabetes	Association between all noise measures and incidence of diabetes 1.08 [1.07-1.09]	1 (low)
Obesity										
Cai et al. (2020)	UK, the Netherlands and Norway	Cohort (longitudinal)	412,934 in UK, 61,032 in the Netherlands and 30,305 in Norway between	Road traffic	Modelled (with measurement data also included) L_{den} in range 42 to 89 dB	Yes including corrections for air pollution (NO2 and PM2.5)	L_{den}	BMI, waist circumference, whole-body fat mass, obesity and central obesity	Associations between L_{den} and obesity markers varied, with significant associations with BMI, waist circumference and obesity in the UK cohort. Stronger associations observed with higher levels of L_{den}. Women with	1 (low)

Publication	Country	Study design	Number of participants	Exposure type	Exposure assessment (method and range)	Confounders considered	Noise measure for relationships	Health endpoints	Reported associations	Risk of bias
			2006 and 2013						lower physical activity, hearing impairment or higher income were more susceptible. Sleep disturbance did not appear to be a significant modifier in the study. Updated WHO meta-analysis based on this and other studies from 2015, including Cramer et al. (2019) and Foraster et al. (2018), with slightly higher pooled estimates but greater variability. Studies included reported variable outcomes.	
Cramer et al. (2019)	Danish Nurse Cohort	Cohort (longitudinal)	15,501 female nurses	Road traffic	Modelled	Yes including corrections for air pollution	L_{den}	Self-reported height, weight, waist circumference and weight characteristics of family members. Calculated BMI and overweight, obesity	No significant associations identified with BMI and waist circumference, no exposure-response relationship identified. Significant effect identified were job strain and degree of urbanisation included. Suggestive of exposure-response relationship for nurses experiencing job strain or living in urban areas.	3 (high)
Foraster et al. (2018)	Switzerland	Cohort (longitudinal)	3796 participants from	Road, rail and	Modelled	Yes including corrections for air pollution	L_{den}	BMI, waist circumference, % body	Significant association between road traffic noise and obesity (RR = 1.26	2 (moderate)

Publication	Country	Study design	Number of participants	Exposure type	Exposure assessment (method and range)	Confounders considered	Noise measure for relationships	Health endpoints	Reported associations	Risk of bias
			SAPALDIA cohort)	Aircraft noise				fat, overweight, obesity and central obesity	[1.05-1.51]). No association with overweight or BMI. For railway noise the results were mixed for obesity. The most significant effects on BMI related to those with self-reported sleep problems. No associations were identified for aircraft noise for any of the indicators evaluated.	
Weyde et al. (2018)	Norway	Cohort (longitudinal)	Focus on children, 6863 where noise exposure occurred during pregnancy, 6403 where noise exposure occurred in childhood	Road noise	Modelled (mean L_{den} = 56.2 dB during pregnancy, Mean L_{den} during childhood decreased over time from 55.4 dB at 18 months to 53.3 at 8 years)	Yes including corrections for air pollution	L_{den}	Length and weight at birth and at various ages, BMI calculated	Road traffic noise during pregnancy showed an association with BMI trajectory in children. For exposures during childhood there was no association with BMI identified.	1 (low)

Cognitive effects and dementia

Table C.7 Review of additional studies – cognitive effects and dementia

Publication	Country	Study design	Number of participants	Exposure type	Exposure assessment (method and range)	Confounders considered	Noise measure for relationships	Health endpoints	Reported associations	Risk of bias
Robinson et al. (2021)	Indonesia	Cross-sectional	48 students	All noise sources in classroom	Measured within the classroom, including aircraft noise (average noise level of 52.92 dB)	Unknown	Not specified, continuous monitor used and average value from 12 measurements presented (time period not stated)	Student concentration	Association between noise intensity of flight and student learning concentration disorders. A regression model was determined for the relationship.	3 (high)
Tzivian et al. (2020)	Germany (Heinz Nixdorf Recall study)	Cohort (longitudinal)	2745 adults (mean age of 64.4 years)	Traffic noise (in general)	Modelled outdoor and indoor levels (mean L_{den} = 53.7 dB, L_{den} (indoors) = 34.74)	Yes, including consideration of air pollution	L_{den} and L_{night}	Cognitive performance assessment, diagnosis of dementia or Alzheimer's disease, intake of cholinesterase inhibitors or other anti-dementia drugs, depressive symptoms	Association between noise (outdoors and indoors) and decreased cognitive function. OR per 10 dB increment were generated, with the variability noted to be lower where indoor noise levels are considered. Threshold for effects noted to be 55 L_{den}, 50 L_{night}, 39 L_{den} (indoors) and 37 dB L_{night} (indoors). Association was stronger for those with severe depressive symptoms.	1 (low)

Publication	Country	Study design	Number of participants	Exposure type	Exposure assessment (method and range)	Confounders considered	Noise measure for relationships	Health endpoints	Reported associations	Risk of bias
Weuve et al. (2020)	US (Chicago)	Cohort (longitudinal)	5227 older adults (>65 years)	Community noise (all sources)	Modelled with measured data also included in model	Yes, including consideration of air pollution	L_{Aeq} (time period unspecified – potentially L_{den})	Mild cognitive impairment (MCI), Alzheimer's disease, cognitive performance and cognitive decline	Association between community noise and MCI and AD as well as worse cognitive performance in older adults. Association relevant to all neighbourhoods (quiet and noisy). Exposure-response relationships identified.	2 (moderate)

Sleep disturbance

Table C.8 Review of additional studies – sleep disturbance

Publication	Country	Study design	Number of participants	Exposure type	Exposure assessment	Confounders considered	Noise measure for relationships	Sleep measures evaluated	Reported associations	Risk of bias
Bartels et al. (2021)	Sweden	Cross-sectional	1764 (women)	Road traffic noise	Modelled and grouped to low (<45 dB), medium (45–50 dB) and high (>50 dB) and considered orientation to façade	Yes, also considered interactions with work stress and effort-reward imbalance (work related)	L_{night}	Self-rated poor sleep (% prevalence)	In general, a non-significant association between noise exposure (where bedroom was facing medium to high traffic) and poor sleep quality was observed. Where the bedroom faced a quiet street, no association identified. Work-related strain also affected self-rated sleep quality. The effect of noise exposure was less than work-related stress.	3 (high)
Basner et al. (2019)	US, Philadelphia	Field study	79 adults	Aircraft traffic	Estimated from noise contours, and measurements in participants room	Yes	L_{Aeq} (sleep period – average was 7.5 hours) and LASmax	Night-time heart rate, body movement, ECG, blood pressure, survey on sleep and health	Statistically significant exposure-response relationship for additional awakenings established for LAmax, where a threshold for effects of 33 dB noted. No statistically significant for blood pressure. For subjective responses, those living near the airport reported poorer sleep quality. Study was small as it was a pilot study. Further field studies are proposed with greater statistical power.	2 (moderate)

Publication	Country	Study design	Number of participants	Exposure type	Exposure assessment	Confounders considered	Noise measure for relationships	Sleep measures evaluated	Reported associations	Risk of bias
Brink et al. (2019)	Switzerland	Cross-sectional	5592 adults	Road, rail and aircraft noise	Modelled at closest façade	Yes	L_{den} and L_{night}, also assessed Intermittency Ratio (IR)	Self-reported sleep disturbance to determine %HSD	L_{night} was found to be a statistically significant indicator of %HSD (crude and adjusted), with aircraft noise having the strongest effect and road traffic noise the weakest, consistent with the outcomes of the review for the WHO. Exposure-response relationships were determined. Noise scenarios with higher IR were associated with higher %HSD. Modifying effect of IR was highest for road noise levels around 60 dB. No difference in exposure-response for those with windows closed, partially open and open. Bedrooms facing away from the road noise source less %HSD. The degree of urbanisation on %HSD was non-significant for all sources.	1 (low)
Di et al. (2019)	China	Cross-sectional	1227	Road and rail traffic noise	Measured and modelled	No	L_{den}	%SD from self-reported survey	Positive association between L_{den} and %SD. %SD found to be higher for males than females in study, with the highest response in individuals 30–40 years of age.	3 (high)

Publication	Country	Study design	Number of participants	Exposure type	Exposure assessment	Confounders considered	Noise measure for relationships	Sleep measures evaluated	Reported associations	Risk of bias
Schreckenberg et al. (2016)	Germany (NORAH study)	Cohort or longitudinal study	3508	Aircraft noise	Modelled or estimated	Yes	L_{den}, L_{night}	Self-reported sleep disturbance to determine %HSD	%HSD decreased significantly after implementation of a night curfew from 11pm to 5am. Exposure-response relationships for degree of sleep disturbances when falling asleep similar before and after night curfew.	2 (moderate)
Skrzypek et al. (2017)	Poland	Cross-sectional	5136 children aged 7–16 years	Road traffic noise	Based on distance of residence to road, not noise levels	Yes	N/A	Prevalence of sleep disorders and attention disorders	Prevalence of sleep and attention disorders higher in most exposed group (<100 m and high traffic density). Exposure-response relationship established.	3 (high)
Trieu et al. (2019)	Vietnam	Cross-sectional	755 adults	Aircraft noise	Measured and modelled	Unknown	L_{night}	Self-reported sleep disturbance as insomnia, measurements of blood pressure and heart rate	Association between L_{night} and %insomnia reported with higher response levels of insomnia reported during the period when night flight operations were enhanced.	2 (moderate)

Table C.9 GRADE review of findings – self-reported sleep disturbance from road, rail, and aircraft noise (%HSD)

Domains	Criterion	Assessment		Grading
		WHO*	Changes from additional studies post WHO**	
Starting point	**Study design: longitudinal = high, others = low**	**Majority cross-sectional**	**Unchanged**	**Low quality**
1. Study limitations	Quality of majority of studies (risk of bias)	All with high risk of information bias	Most with moderate to high risk of bias	Downgrade one level, unchanged with new studies
2. Inconsistency	Conflicting results	Inconsistent, variable results	Unchanged	Downgrade one level, unchanged with new studies
3. Directness	Direct comparison; same PECO	Same PECO	Unchanged	No downgrade
4. Precision	Small sample size or Low numbers of events (HA) OR wide confidence intervals	CI narrower than 25%	Unchanged	No downgrade
5. Publication bias	Funnel plot indicates	Not able to be assessed	Not reviewed	No downgrade
Overall judgement				Very low quality
6. Exposure assessment	Statistically significant trend	Yes	Yes for most studies	Upgrade
7. Magnitude of effect	Fit of logistic regression (RR > 2)	RR > 2 for road and rail	Unchanged however some of the new studies show lower RR	Upgrade (potential for no upgrade)
8. Confounding adjusted	Effect in spite of confounding working towards the nil	Not observed	Unchanged	No upgrade
Overall judgement				**Low to moderate quality**

* WHO evaluation of studies as presented by Basner et al. (2018)

** Additional studies reviewed by DEFRA (van Kamp et al., 2020) and in this assessment

Other potential effects

Table C.10 Review of additional studies – birth and reproductive effects and cancer

Publication	Country	Study design	Number of participants	Exposure type	Exposure assessment (method and range)	Confounders considered	Noise measure for relationships	Health endpoints	Reported associations	Risk of bias
Birth and reproductive										
Weyde et al. (2018)	Norway	Cohort (longitudinal)	Focus on children, 6863 where noise exposure occurred during pregnancy, 6403 where noise exposure occurred in childhood	Road noise	Modelled (mean L_{den} = 56.2 dB during pregnancy, Mean L_{den} during childhood decreased over time from 55.4 dB at 18 months to 53.3 at 8 years)	Yes including corrections for air pollution	L_{den}	Length and weight at birth and at various ages, BMI calculated	Road traffic noise during pregnancy showed an association with BMI trajectory in children. For exposures during childhood there was no association with BMI identified.	1 (low)
Argys et al. (2020)	US (New Jersey)	Cross sectional	Birth records between 2004 and 2016	Aircraft noise	Measured in 2014 and extrapolated for study period (mean aviation noise of 47.153 dB, mean aviation and road noise 49.949 dB)	Yes including corrections for air pollution	L_{den}	Low birth weight	Increase of 1.6% in likelihood of low birth weight for mothers living close to the airport, in the direction of the runway, exposed to noise above 55 dB and during the period of higher airport activity.	2 (moderate)

Publication	Country	Study design	Number of participants	Exposure type	Exposure assessment (method and range)	Confounders considered	Noise measure for relationships	Health endpoints	Reported associations	Risk of bias
Cancer										
Sorensen et al. (2021)	Denmark	Cohort (longitudinal)	Population model	Road and railway noise	Modelled (L_{den} max and L_{den} min) road traffic noise at the most exposed and least exposed façades of buildings for 1995, 2000, 2005, 2010 and 2015, with linear interpolation for years in range 1990 – 2017. Rail noise modelled at most exposed façades for 1997 and 2012 and interpolated. Models were validated	Yes including corrections for air pollution, however lifestyle factors not addressed	L_{den} (max and min)	Incidence of breast cancer reported in Danish Cancer Registry (as well as data on ER and HER2)	Road traffic noise was associated with increased risk for breast cancer, especially at the least exposed façade. Exposure to railway noise also seemed associated with a higher rate of breast cancer. Exposure-response relationships determined from the study. The stronger association with the least exposed façade supports the theory that noise during sleep is a critical exposure period in relation to breast cancer.	1 (low)

Table C.11 GRADE review of findings – aircraft noise and incidence of low birth weight

Domains	Criterion	Assessment		Grading
		WHO*	Changes from additional studies post WHO**	
Starting point	**Study design: longitudinal = high, others = low**	**Case-control studies**	**1 additional case-control study**	**Low quality**
1. Study limitations	Quality of majority of studies (risk of bias)	Yes	additional study has moderate risk of bias	Downgrade one level, unchanged with new study
2. Inconsistency	Conflicting results	Similar direction but overall unclear	Unchanged	Downgrade one level, unchanged with new study
3. Directness	Direct comparison; same PECO	Yes direct evidence	Unchanged	No downgrade
4. Precision	Confidence interval contains 25% harm or benefit	Only for one study clearly	Unchanged	Downgrade one level, unchanged with new study
5. Publication bias	Funnel plot indicates	Not able to be assessed	Not reviewed	No downgrade
Overall judgement				Very low quality
6. Exposure assessment	Statistically significant trend	Only for one study clearly	Suggested for new study, for specific situation	No upgrade
7. Magnitude of effect	Fit of logistic regression (RR > 2)	No	Not determined	No upgrade
8. Confounding adjusted	Effect in spite of confounding working towards the nil	Not fully adjusted	Not fully adjusted	No upgrade
Overall judgement				**Very low quality**

* WHO evaluation of studies as presented by Nieuwenhuijsen et al. (2018)

** Additional study identified in this review

Appendix D: Glossary

Term, phrase, or acronym	Explanation
%HA	Percentage of the population highly annoyed
%HSD	Percentage of the population highly sleep disturbed
AADT	Annual average daily traffic
AC-10	Asphaltic concrete
AMI	Acute myocardial infarction
ArcGIS Pro	GIS software application
CI	Confidence interval
CM	Choice modelling
CNOSSOS-EU	Common Noise Assessment Methods in Europe
Cost model	Model prepared using Microsoft Excel to calculate social health costs of noise using the noise prediction data from the noise model and exposure-response relationships from the health review.
CRN	UK Department of Transport Calculation of Railway Noise
CRN	Calculation of Rail Noise
CRTN	Calculation of Road Traffic Noise
CV	Contingent valuation
CWR	Continuously welded rail
DALY	Disability-Adjusted Life Year
dB	Decibels
dBA	A-weighted decibels
DEFRA	Department for Environmental, Food and Rural Affairs (UK)
DEM	Digital Elevation Model
DSM	Digital Surface Model
DW	Disability weight
END	Environmental Noise Directive
EU	European Union
FNM	Façade noise map
GDE	Gross domestic expenditure
GDP	Gross domestic product
GIS	Geographic Information System
GNM	Grid noise map
GRADE	Grading of Recommendations, Assessment, Development and Evaluation system
GVA	Gross value added
HAPINZ	Health and Air Pollution in New Zealand
HP	Hedonic pricing
HRQOL	Health Related Quality of Life
ICB	Inner Control Boundary

Term, phrase, or acronym	Explanation
ICD	International Classification of Diseases
IHD	Ischaemic heart disease
$L_{10,T}$	Centile noise level equalled or exceeded for 10% of a given time period T.
$L_{A10,18h}$	A-weighted centile level equalled or exceeded for 10% of an 18-hour time interval. This is the noise descriptor used for interpreting road traffic noise in accordance with the Calculation of Road Traffic Noise (CRTN) prediction method.
L_{AE}	A-weighted Sound Exposure Level.
$L_{Aeq,18h}$	A-weighted equivalent continuous sound pressure level during an 18-hour period expressed in dB. This is the basic noise descriptor used for rail noise in the Calculation of Rail Noise (CRN) prediction method.
$L_{Aeq,24h}$	A-weighted equivalent continuous sound pressure level during a 24-hour period expressed in dB. The preferred noise descriptor used for road traffic noise assessment in New Zealand in accordance with New Zealand Standard 6806:2010 *Acoustics – Road Traffic Noise – New and Altered Roads* (NZS 6806). A correction of -3 dB is applied to $L_{A10,18h}$ for conversion to $L_{Aeq,24h}$.
L_{Amax}	Maximum sound level measured during a measurement period, A-weighted.
L_{ASmax}	A-weighted, slow response, maximum, sound level.
L_{day}	Equivalent continuous sound pressure level when the reference time interval is the day (07:00–19:00). A component to calculate L_{den}. Also known as $L_{Aeq,12h}$
L_{den}	A-weighted over a 24-hour period that includes a penalty of the evening and night-time level to account for the elevated noise sensitivity in these time periods (as defined in section 3.6.4 of ISO 1996-1:2016).
L_{dn}	The LAeq over a 24-hour period with a penalty of 10 dB for noise during the hours of 23:00–07:00.
$L_{eq,T}$	Equivalent continuous sound pressure level during a given time period T.
$L_{evening}$	Equivalent continuous sound pressure level when the reference time interval is the evening (19:00–23:00). A component to calculate L_{den}. Also known as $L_{Aeq,4h}$
LIDAR	Light Detection and Ranging – method to determine ranges in map making.
LINZ	Land Information New Zealand
L_{max}	Maximum sound level measured during a measurement period.
L_{night}	Equivalent continuous sound pressure level when the reference time interval is the night (23:00–07:00). A component to calculate L_{den}. L_{night} is widely used for exposure assessment in health effect studies.
LTNZ	NZ Transport Agency Waka Kotahi
NDI	Noise Depreciation Index
Noise model	Model prepared in SoundPLAN 8.2 to calculate noise from New Zealand's road (state highways and arterials) and rail transportation network.
NSDI	Noise Sensitivity Depreciation Index
NZSoS	New Zealand School of Surveying
OCB	Outer Control Boundary
OSM	Open Street Map
PAF	Population attributable fraction
PPF	Protected Premises and Facilities
QALY	Quality-Adjusted Life Year

Term, phrase, or acronym	Explanation
RAD	Restricted activity day
RMA	Resource Management Act
SA1	Statistical Areas Level 1
SEL	Sound Exposure Level
Shapefile (.shp)	File format for storing geometric and attribute information of geographic features.
SoundPLAN	Software for calculation of noise
TIN	Triangular Irregular Network, a datatype used in GIS
TRL	Transport Research Laboratory
VoLY	Value of a statistical life year
VoSL	Value of a statistical life
VPF	Value of preventing a statistical fatality
WHO	World Health Organization
WTA	Willingness to accept
WTP	Willingness to pay