Ref. Ares(2021)2866764 - 29/04/2021

Noise and **E**missions
MOnitoring and Radical Mitigation

Research and Innovation Action

H2020- LC-MG-1-9-2019

Methodology for external cost estimations

WP8, Task 8.1.1

[Version 1, 29-04-2021]

Author(s): David Birchby (Ricardo), Sandra Fischer (Ricardo), Louisa Kramer (Ricardo), Andrew Lewin (Ricardo), Nicola Masey (Ricardo), Jacob Sendall (Ricardo), Scott Hamilton (Ricardo)

www.nemo-cities.eu @NemoProjectEU

This project has received funding from the European Union's Horizon 2020 research and innovation programme under the Grant Agreement n° 860441.

Disclaimer

The content of this deliverable reflects only the author's view. Neither CINEA nor the European Commission is responsible for any use that may be made of the information it contains.

Copyright notice

©2020 NEMO Consortium Partners. All rights reserved. NEMO is a HORIZON2020 Project supported by the European Commission under contract No. 860441. For more information on the project, its partners and contributors, please see the NEMO website (www.nemo-cities.eu/). You are permitted to copy and distribute verbatim copies of this document, containing this copyright notice, but modifying this document is not allowed. All contents are reserved by default and may not be disclosed to third parties without the written consent of the NEMO partners, except as mandated by the CINEA contract, for reviewing and dissemination purposes. All trademarks and other rights on third party products mentioned in this document are acknowledged and owned by the respective holders. The information contained in this document represents the views of NEMO members as of the date they are published. The NEMO consortium does not guarantee that any information contained herein is e-free, or up-to-date, nor makes warranties, express, implied, or statutory, by publishing this document.

Document Information

Grant agreement	860441
Project title	Noise and Emissions MOnitoring and Radical Mitigation
Project acronym	NEMO
Project coordinator	Mª Dolores Hidalgo, PhD (dolhid@cartif.es) Fundación CARTIF.
Project duration	May 2020 – Apr 2023
Related work package	WP8: Impact Analysis
Related task(s)	Task 8.1: Externalities and Charging Methods
Lead organisation	RIC – Ricardo
Contributing partner (s)	
Reviewers	ORSE – OPUS RS EUROPE S.L
	T&E – European Federation for Transport and Environment
Due date	30th April 2021
Submission date	29th April 2021
Dissemination level	Public

History

Date	Version	Submitted by	Reviewed by	Comments
15/04/2021	0.1	Louisa Kramer	-	Draft version
29/04/2021	1	Louisa Kramer	T&E; Javier Buhigas (ORSE)	Final Version

Table of content

List of figures

List of tables

Abbreviations and acronyms

Acronym	Description
AADT	Annual Average Daily Traffic
ANPR	Automatic Number Plate Recognition
ATC	Automatic Traffic Count
CFD	Computational Fluid Dynamics
COPERT	COmputer Programme to calculate Emissions from Road Transport
CRF	Concentration Response Function
DALY	Disability-Adjusted Life-Years
dB	Decibel
GIS	Geographic Information System
GPS	Global Positioning System
HE	High Emitters
HGV	Heavy Goods Vehicles
IPA	Impact Pathway Approach
LIDAR	Light detecting and ranging
MGE	Mean Gross Error
PM	Particulate Matter
N-RSD	Noise emissions Remote Sensing Device
RMSE	Root Mean Squared Error
RSD	Remote Sensing Device
QALY	Quality-Adjusted Life Years
WTP	Willingness To Pay
YLD	Years of Life with Disability
YOLL	Years of Life Lost

Executive Summary

This document is part of the wider implementation of Mobility for Growth which is driving the Focus Area: "Building a low-carbon, climate resilient future". Its focus lies on low carbon and sustainable transport and on upgrading transport infrastructure to monitor noise and emissions.

Pollutant and noise emissions can have a detrimental impact on human health. Breathing in pollutants such as particulate matter (PM) can have effect on lung function and exacerbate symptoms in individuals with underlying respiratory diseases. Prolonged exposure to elevated noise levels can also cause health issues, such as sleep disturbance, increased stress, and cardiovascular risks.

Within major cities, the transport sector is one of the largest contributors to noise and pollutant emissions. The Noise and Emissions Monitoring and radical mitigation (NEMO) goal is to develop a solution for the integration of autonomous systems into existing infrastructure in order to measure and mitigate emissions and noise levels. The new system, alongside the implementation of new mitigation measures, aims to provide a solution to improve air quality and reduce noise impact in European cities. The technology developed will provide measurements of noise and emissions from individual vehicles on the road, or individual trains, in order to identify high-emitters (HEs).

It is important to assess the impact of the innovative technologies implemented in NEMO on human health and the environment to compare to the costs of novel techniques and systems designed to mitigate these impacts. The solutions will carry external costs and benefits associated with pollution and noise emissions. To compare the monetarily cost to benefits, an approach to monetise the environmental benefits is required.

Within this report, a review of methodologies to assess the impact of pollutant and noise emissions on human health and the environment is provided. Based on this review and the integration of remote sensing measurements of pollutant and noise emissions, undertaken as part of the NEMO project, recommended methodologies for calculating external costs linked to emission and noise are given.

Keywords

Emissions, externalities, damage costs

1 Introduction

1.1 Purpose, scope and target group

This document provides a review of damage cost information, based on the most recent evidence and guidance documentation across the EU, and a recommended methodology to calculate external costs from emission and noise incorporating data that will be obtained by the enhanced remote sensing device (RSD), developed as part of NEMO.

The cost information resulting from this methodology will form the basis for the policy approach in which the impact of noise and emissions reducing measures should bring a reduction of at least 30% in emissions and 20% in noise in targeted zones based on the measured level at the beginning of the project. A flexible approach is provided, which can be applied to different scenarios to appraise the potential impacts of a policy.

1.2 Contribution partners

Table 1-1: Contribution of partners

Partner nº and short name	Contribution
T&E	Leading WP 8: Impact Analysis
RIC	Leading Task 8.1.1: Estimate external costs associated with pollutant and noise emissions
ORSE	Assist RIC in the methodology for calculation of externalities
OTS	Assist RIC in the methodology for calculation of externalities
JRC	Support for WP 8

1.3 Relation to other activities in the project

Table 1-2: Relation to other activities in the project

Task	Description
8.1.2	Transport policy options identified and assessed D8.2: Methodology for considering external costs in charging, access & information policies
8.1.3	White paper written and published that proposes best practises to deliver the 30% improvement in AQ and 20% reduction in noise. D8.3: Whitepaper: charging methods based on RDE
8.1.1	Economic evaluation of mitigation scenario D8.5: Economic evaluation of mitigation scenarios

2 Overview and high level approaches

External costs relate to the costs of an activity upon an individual or society. Air pollution costs and noise costs are two of the key external costs associated with changes in road transport. Therefore, it is important to consider these costs when making policy decisions relating to the transport sector. Air pollution and noise emissions are also effects for which methodologies to appraise and monetise the impacts have been most matured, specifically the dose-response-relationship between the exposure of air pollutants and the associated health risks.

The emission of air pollutants can lead to different types of externalities or damages:

- Health effects: The inhalation of air pollutants such as particles (PM_{10}, $PM_{2.5}$) and nitrogen oxides (NOx) leads to a higher risk of respiratory and cardiovascular diseases (e.g. bronchitis, asthma, lung cancer). Nitrogen oxides are also involved in the formation of ozone (O_3), a secondary pollutant which can irritate the eyes, and have adverse impacts on health when breathed in. These negative health effects lead to medical treatment costs, production loss at work (due to illness) and, in some cases, even to death. The impact on health will be the main area of focus in this methodology.

- Material and building damage: Air pollutants can mainly lead to two types of damage to buildings and other materials: a) pollution of building surfaces through particles and dust; b) damage of building facades and materials due to corrosion processes, caused by acidic substances (e.g. NOx or sulphur dioxide (SO_2)). Less focus will be given to those effects in this methodology.

- Crop losses and biodiversity loss have less importance in the methodology described here, due to the focus on city centres.

- Indirect economical externalities: health damage has an effect on job absenteeism or other forms of reduction of a territory productivity. The increase of premature deaths has a similar effect. In addition, an improvement on air quality has an effect on the quality of life of a city, which can increase tourism or retail business.

Noise emissions can lead to the following impacts:

- Health effects: Long-term exposure to excess noise can lead to increased stress, hypertension, and cardiovascular risks (myocardial infarction).

- Lifestyle effects: Sleep disturbance and annoyance.

External costs may be determined as average external costs or marginal external costs. While average external costs are similar to total costs, marginal external costs are the additional external costs that occur due to an additional activity. The units the costs in both cases are presented in are usually €-cent/p-km for passenger, €-cent per t-km (tonne) for freight or €-cent/veh-km (vehicle) for transportation quantity.

There are a range of approaches and guidance to appraise external costs associated with pollutants and noise from transport. In the following sections a review of these methodologies is given. Pollutant emissions and noise are addressed separately as there are different approaches associated with these two externalities. Based on the review and official guidance recommended methodologies for calculating external costs are presented.

3 Air Quality Modelling of Road Traffic Sources

3.1 Introduction

It is useful to start with some assumptions as to the general scope of air quality modelling that might be employed following on from or as part of this project. Setting these out will provide the technical context for much of the methodological recommendations we provide later. We assume that:

1. The air quality models are likely to involve **urban settings**, and cover **reasonably large spatial extents** (up to around 50km x 50km or so)
2. The **output data will be used in subsequent analyses** that could require a range of output data formats such as numerical output files (e.g. .csv), GIS datasets (e.g. .shp, .tif). For example, economic assessment, health impact analysis or other geospatial analysis
3. The models used will need to represent a **wide variety of road traffic source types**, such as gasoline/diesel cars, trucks and buses and how they behave in the domain (speed, flow, temporal variation). This mostly affects the emission model and to a lesser extent the dispersion model
4. The emission model should be able to **accommodate custom emission rates** for vehicles that have been measured in situ during remote sensing campaigns (this is discussed further in Section 3.4.6)
5. The models will need to **place the sources in the correct geospatial context** alongside sensitive receptor locations
6. The models should produce **outputs in convenient data formats** to allow subsequent analysis. Most likely these will be in the form of numerical predictions in spreadsheet or text files, and GIS data formats such as shapefiles or rasters).

Given the assumptions above, we can now discuss the technical matters arising which would shape any air quality modelling studies that are carried out after this project.

3.2 Benefits and limitations of air quality models

Historically, air quality assessment has been based on monitoring data, as this is considered to be as close to reality as is possible. Even though modelling is often seen as being more uncertain than monitoring, there are three major reasons for using models in combination with monitoring for air quality assessments.

- The spatial coverage of monitoring is usually limited. Modelling can potentially provide complete spatial coverage of air quality.
- Modelling can be applied prognostically, i.e. it can be used to predict the air quality as a result of changes in emissions or changing meteorological conditions.
- Modelling provides an improved understanding of the sources, causes and processes that determine air quality

There are, however, a number of limitations attached to air quality models:

- They require extensive input data particularly in relation to emissions and meteorology.

- They remain uncertain in their predictions so validation with monitoring data is usually always required
- Their ability to represent the real world is limited as regards spatial resolution and process descriptions, for instance. Models remain a simplified representation of reality.
- Effective and quality-controlled modelling requires expert users.

3.3 Use of models in air quality assessment

European countries have not routinely employed a unified approach to air quality modelling. This has resulted in a range of models being applied, at both national and local level, in various forms. Whilst the models being used in Europe are varied according to the situation, it is expected that models be comparable, well documented, and validated for their required applications to achieve reliable modelling results. This applies to the type of modelling that is implied in this project- urban scale dispersion modelling of road transport emissions.

Models applied for assessing the existing AQ situation in an area (like a city) aim at estimating concentrations of criteria pollutants (such as NO_2, PM_{10}, $PM_{2.5}$, CO), calculating population exposure and health impacts, and identifying air pollutant source contributions. In the case of NO_2, it is common to model NOx and convert the resulting concentrations to NO_2 in post-process.

Air quality models are rarely used without any reference to measurement data for validation purposes, i.e. will compare modelled and measured results (often using statistical methods to demonstrate performance). Modelling results are particularly useful in assessments to provide additional concentrations for the geographical area not extensively covered by measurement data. Modelling can be further applied to calculate population exposure at concentration levels above, for example, limit values.

3.4 Operational air quality modelling of road traffic sources in European cities

There are a wide variety of air quality models in routine use by environmental agencies and other practitioners around the world. These range from conceptually and computationally simple semi-empirical Gaussian plume models, via moderately complex 'puff' models, to very demanding numerical models based on computational fluid dynamics (CFD). It follows that there are a range of modelling tools based on these schemes that are designed for the purpose of modelling air pollution arising from road traffic. Deciding on which air quality model to use for a given study is normally a balance between sophistication of the system and the time and resources available to undertake the assessment. The aim is to find the right balance of these two counteracting aspects.

Most air quality modelling studies which are carried out in an operational (non-research/non-academic) setting need to be carried out under constraints on time and/or resources. Whilst it may be desirable to use systems that utilise, say, the 'best' treatment of urban physics, these are usually too slow to be operationally useful- especially given the air quality standards being assessed against usually require annual simulations (8760 discrete hours). The most complex systems are typically used in research institutes or for small-scale projects involving a few streets in a city with little evidence of widespread application at the city scale.

Modelling the air quality impacts of road traffic presents us with a set of special problems which must be considered as part of any operational simulation.

- Sources and domain size- road traffic networks are very complex in that they are usually spatially represented as a group of discrete road links, each with their own traffic/emissions/geometry. Such a road network will typically cover a large spatial area and we may wish to treat this with a high degree of spatial fidelity, which is another crucial parameter in determining computation time. These effects compound in a typical city scale (say, up to 50km x 50km) simulation and can result in computational time that is measured in days, or weeks for a single simulation. Usually this is operationally prohibitive, so steps need to be taken to improve the simulation time, whether this be through using a simpler or more optimised model, or handling the domain differently (e.g. by splitting across different runs, making it less spatially detailed). Even in cases where a domain can be split into parts, this does not usually bring about efficiencies that suddenly make the most complex modelling schemes viable.

- Complexity- models can take various forms, from simple to advanced. This continuum is reflective of the complexity of the numerical/physical schemes used in the models. The more complex/complete the physics, the greater number of calculations the host computer must make, which means that run time follows the same trajectory. To our knowledge there are no 'complex' and 'fast' air quality models (e.g. there is no CFD that can diagnose concentrations at scale in a city in a sensible amount of time). Generally, using models with a more 'complete' treatment of the physics of dispersion in the urban setting leads to run times that are operationally prohibitive, especially when operating at scales beyond a few streets in a city. There is an obvious trade-off between trying to represent the 'true' physics of dispersion in city environments (which takes a very long time), versus the implications for operational use of the modelling system (which need results in a sensible amount of time). This means that most agencies recommend air quality models that do not treat the physics of dispersion in the 'true' sense, but rather treat dispersion in a simpler manner but still retain the most important characteristics governing the relationship between the source emission strength and the concentration experienced by a receptor. The use case for the model outputs also tends to inform the level of complexity that can be accommodated in the modelling system- for example if broad population average concentration estimates are suitable, say for health impact calculations, a simpler model could be used. For cases that are very specific to a given urban form, say a building is to be reconfigured, a more complex model can normally be applied but in a very spatially confined domain.

  CFD models are, to our knowledge, never used in an operational setting, probably due to their current limitation to idealised, stationary and very fine scale applications - mainly arising from their intense computational requirements. It is not currently possible to calculate annual statistics without resorting to statistical inference based on short term results from CFD. Hence most practitioners tend to avoid using them for regulatory applications and such models are normally only used for micro-scale assessments involving very complex flows around buildings e.g. at industrial facilities.

- Usability- in general, dispersion models are considered more accurate as the models become more fundamental and hence computationally more intensive. However, the latter types of models have their own pitfalls and require a great deal more expertise and computational / institutional resources to run. Hence, higher accuracy is not guaranteed simply due to choosing a more 'sophisticated' model- complex systems are prone to user error or poor simulation design which can negate the benefits of choosing a more complex model. In practice there is a balance to be struck where a model is chosen that best suits the needs of the study, the available computational resources, and the skill of the user.

Given the assumed scope of air quality modelling projects we set out, and in light of the “special problems” we outlined, it is likely that the most practical/feasible models for this context are Gaussian plume models which are specially configured to treat road traffic emission sources. Gaussian models belong to the Eulerian category of models - that is they use a fixed coordinate system during computation. These types of models are typically used to model the air pollution effects of single or small groups of sources in the near field- usually for source/receptor distances up to about 50km.

It is important to note that there is no standard reference air quality model that serves all types of source and all spatial scales. In practice, most agencies use a combination of models for different purposes. This document does not attempt to review or discuss the full range of air quality models that are used for other such purposes.

3.4.1 Tools for modelling air quality in urban settings

We have described a range of general technical matters which are relevant to the air dispersion modelling activities which will be undertaken in cities as part of this program. Even with these technical considerations there remains a variety of air quality models which could be deployed for road traffic sources in European cities. The models that we suggest below would all be useful candidates for practical application by agencies involved in this initiative in future. That said, we do not make specific recommendations as to the exact model that is recommended for use. Rather, our candidate list is focused on providing the minimum recommended features that the system chosen by the modelling group should include. In doing so our aim is not to prejudice the choice of one modelling system over any another, provided the models under consideration meet the suggested requirements. The list below should not be considered exhaustive and there are likely other systems which could be deployed whilst offering a similar balance of functionality and operational efficiency.

Therefore, the list of models shown below – whilst not being exhaustive – aims to provide a useful ‘sample’ of available air quality models that we have experience of, and which could be applied to the urban road traffic context. It will be for the modeler to select their preferred air quality model in each study but there are some general 'fit-for-purpose' criteria that should apply:

- The model has the appropriate spatial and temporal resolution for the intended application- in this case the ability to operate in annual average simulations.
- Receptor locations can be selected that are spatially located in three dimensions (x, y, z, expressed in metres).
- The model is adequately validated for the application, either in model documentation, in studies where it has been applied or in scientific literature.
- The model contains the relevant physical and chemical processes suitable for the type of application, scale and the pollutant for which it is applied.
- The relevant emission sources for the application are adequately represented. In this case it must be able to represent discrete emission sources for many hundreds, perhaps thousands of road links in an urban domain.
- Meteorological data are available to drive the model, which should be complete (free from major gaps) and broadly representative of the area in the study.

All the models described below meet these criteria and undoubtedly other models we are less familiar with will meet them as well.

3.4.1.1 AERMOD

AERMOD is a sophisticated Gaussian plume model based on continuous parameterization of atmospheric dispersion. AERMOD is the preferred model of the USEPA for line, area, volume and point sources and is mandatory for road traffic sources under most circumstances. AERMET is the meteorological pre-processor of AERMOD [1] [2]. These models have greatly increased in importance in recent years, as EPA decided in 2005 to replace ISC3 by AERMOD as the preferred air dispersion model. AERMOD and AERMET comprise what is probably the most widely used dispersion modelling system in the world.

AERMOD consists of a meteorological interface and an actual dispersion model. The meteorological interface consists of the calculation of the wind speed profile, the potential temperature profile, and the profile of the vertical and lateral dispersion parameters. The actual dispersion model covers the calculation of the concentration and includes issues such as plume rise, building downwash, plume dispersion modifiers, and plume meandering effects.

AERMOD, including the pre-processor AERMET, is freely available on the USEPA website at http://www.epa.gov/ttn/scram/dispersion_prefrec.htm. Source code, executable file, documentation, and extensive supporting material can be downloaded there. At the time of writing the most recent version of the AERMOD system is version 19191.

AERMET is the meteorological pre-processor of AERMOD. The function of AERMET is to calculate the heat balance of the surface, determine whether the atmosphere is stable or convective (i.e., whether the heat flows from the atmosphere to the surface or vice versa), and calculate the friction velocity and the convective velocity scale, as well as the Monin-Obukhov length.

AERMOD requires the following information when modelling an area source using the LINE source keyword (as used in this work):

- The emission rate per unit area (mass per unit area per unit time);
- The coordinates of midpoint of the ends (X1, Y1, X2,Y2);
- The width of the source in metres;
- The initial vertical dimension of the area source plume and initial vertical dispersion coefficient; and
- The release height above the ground.

When the emission sources have been set up, AERMOD requires a list of receptor locations and meteorological files from AERMET to execute the simulation.

The AERMOD model includes a LINE source type which can be used for road traffic sources. The model can accommodate an unlimited number of road sources in a single model run- though the run time can become problematic beyond a few thousand links. The LINE source allows the road width to be set, say for a width of 7m in a common two-lane carriageway. All emissions on the links are pre-calculated for the area in question so the main task is to import these geometries and emissions accurately and reproducibly into AERMOD. Most emission models provide outputs per unit of road length (g/km), and usually with a time component (g/km/s) to allow integration with the air quality model. For AERMOD the emission has to be first transformed to area source terms, which is done by dividing the length-based emissions by the road width.

The sources in AERMOD should be modelled using the 'urban' switch in the control pathway and using diurnal factors (expressed in the EMISFAC section of the aermod.inp control file) to represent time varying emissions.

AERMOD requires an estimate of the population of the urban area and we set this to 250,000 based on available census data for the area. In our experience concentrations from the AERMOD model not very sensitive to this parameter, so broad population estimates can be used. All receptors use a height of 1.5m above ground.

We used another python script to import the list of receptor locations, which were common to all the models under test. The scripts we developed to create the file that controls the AERMOD model and the external files which set source and receptor parameters have been supplied to JAQU alongside this report. We also provide the modelling framework including AERMOD code, receptor files, source files and meteorological data used in the work.

3.4.1.2 ADMS-Roads

ADMS-Roads is a steady state quasi-Gaussian dispersion model developed in the UK by CERC, following an initial collaboration and funding in 1990, involving a number of Government agencies and others (including the Met Office, power generators, HMIP- the forerunner of the Environment Agency- and the University of Surrey).

It covers dispersion from point, area, volume and line sources with a straight-line plume trajectory from source to receptor or grid point. Concentrations are modelled on an hour by hour basis using an appropriate (usually) regional meteorological data set. The model is commercially available in a number of permutations: ADMS-Urban, which essentially nests the point, area and volume model ADMS 5 and the line source model ADMS-Roads into one package, while ADMS-Airports is suitable for multiple sources on an airport. The point source component of the model takes account of plume rise and building downwash. There are options to apply diurnal, weekly and monthly profiles to the emissions or detailed hour-by-hour profiles for a full year.

While the use of ADMS-Urban in the UK has been relatively limited, the ADMS 5 and ADMS-Roads models are widely used and there is a strong user community with experience of using ADMS models.

The line source component is used to model open roads, with a separate module for dispersion within street canyons, based on the Danish OSPM model.

The ADMS-Roads model uses much the same emissions, meteorology and source geometry definitions as the other models described here. The main difference arises from the limitations on the number of road links that can be modelled in a single instance of the ADMS-Roads license- a few hundred road links is the upper maximum. This presents quite a challenge for city scale modelling of the type implied in this work which can involve many thousands of road sources. Usually to work around the limit in road sources a modeler will split their domain into several parts though this can require multiple licenses to work which may not be practical in this context.

As with the other models, the road width is set in the model (say, 7m) in the input files and can be set for each individual link.

3.4.1.3 RapidAIR

A technical description of the RapidAIR model is available in [3]. A summary of key functionality is provided below.

RapidAIR is an air quality modelling system that has been developed as a decision support tool to aid evidence-based management of air quality in cities. The system comprises of the following component parts;

- A meteorological data acquisition, processing, filling and modelling module based on the USEPA AERMET code and data handling methods sourced from international protocols.
- A road traffic emissions module based on coefficients from the COPERT5 database, which calculates emissions across entire road networks in a few seconds.
- A road traffic emissions dispersion model based on the USEPA AERMOD code, which uses a computationally efficient convolution based technology to produce results.
- For UK studies there is a module which automatically gathers, processes and integrates background air pollution data from the UK and devolved government's data.
- A street canyon model based on the UK Met Office model AEOLIUS [4], which is closely related to the OSPM model. This includes an automatic canyon allocator, which uses a building height dataset to establish geometries in any domain in the world.
- Functionality to interrogate concentrations at any location in a domain (which can comprise several hundred million discrete predictions).
- Plotting of key meteorological and spatial datasets.

The system is based on existing and well-accepted and validated emissions, meteorological and dispersion models. RapidAIR uses the AERMOD dispersion model from the USEPA to disperse road traffic emissions. AERMOD is an accepted model for this purpose as set out by the USEPA in their statutory technical guidance for dispersion modelling of road traffic emissions. Thus we can expect a high level of agreement between the RapidAIR and AERMOD models (which is indeed what we found in this work).

The RapidAIR system includes a high spatial resolution dispersion model (1m or greater to characterise near-field concentration gradients along roads. The road dispersion model uses an area source dispersion kernel method which is parameterised according to USEPA recommendations. The kernel is created in AERMOD for unit emission and the results are then scaled according to the emission intensity distributed along the line sources representing the roads.

The emission intensity is presented to the model as a grid, which is based on calculations of emissions in the emissions module of RapidAIR (based on COPERT 5) as a function of traffic flows, speed, fleet mix and fuel use. The model is set to provide period average results, which can be any time interval from one hour to one year. The user can tune several of the kernel parameters, such as size, resolution, release height. The road source concentrations can be easily added to the outputs of other dispersion models for point or area sources as the meteorology is the same.

3.4.1.4 R-Line

The R-Line model is a research dispersion modelling tool under development by the US EPA's Office of Research and Development for linear sources. The model is based on a steady-state Gaussian formulation and is designed to simulate line-type source emissions by numerically-integrating point source emissions [5]. R-Line is designed to simulate primary, chemically inert pollutants with emphasis on near surface releases and near source dispersion. The concentration from a finite line source in R-Line is found by approximating the line as a series of point sources. The number of points needed for convergence to the proper solution is determined by the model and is a function of distance from the source line to the receptor. Each point source is simulated using a Gaussian plume formulation. The model has several features that distinguish it from other models. It includes new formulations for the vertical and horizontal plume spreads of near surface releases based on historical field data, a recent tracer field study and recent wind tunnel studies. It is also able to simulate vertical and lateral dispersion rates and low-wind meander conditions.

To facilitate application of the model, its meteorological inputs are consistent with those used by the AERMOD model and simplified road-link specifications. The specific variables that are needed by R-Line include the surface friction velocity, the convective velocity scale, Monin-Obukhov length, the surface roughness height, and the wind speed and direction at a reference height within the surface layer. Additionally, for light-wind stable conditions when the surface friction velocity is generally small, an adjustment is made to the friction velocity based on [6].

The model setup requires the compilation of R-Line code (i.e. the generation of executable files) that match the architecture of the machine that will be running them. Once the model has been successfully compiled, the setup is a relatively simple task, as text files are used to control the sources, meteorology, receptors and programme run options.

The first step is the generation of the “Source” and “Receptor” files; the Source file is a text file that contains each of the roads to be modelled, represented as one source per line. The geometry of the roads can be defined through start and end points for each of the road or the endpoints of the centre of a group of sources and an offset distance for each source relative to the centreline.

Additional parameters that need to be provided by the user are the initial vertical dispersion ($\sigma z0$, in meters) and the number of lanes for each specified road link. Road width is set similarly to the other models.

According to the model documentation, the selection of the initial vertical dispersion should be set based on the average vehicle height. The emissions also need to be defined in the Source file, provided in units of g/m/s. The next step is to define the Receptor file, which is a text file with geographic coordinates (Easting, Northing and Height) of the points at which the user wants to extract concentrations. The interaction between the user and the model is done through an “Input” file, which defines the names of the Source and Receptor files, the meteorology to be used, and the set of concentrations that need to be calculated (plume, meander, or plume + meander). It also enables the user to specify the temporal aggregation of the results (monthly or hourly).

3.4.2 Spatial resolution of the urban air quality model

The required resolution generally varies depending on the pollutant and on the type and scale of the assessment. In this context it is assumed that the modelling will be focused on long term air quality standards for NO_2 and PM_{10}/$PM_{2.5}$ at the scale of a typical city.

From a modelling perspective, the following recommendations concerning resolution should be followed:

- The modelling should focus on (and perform best) at sites where the concentrations are likely to be highest, e.g. the kerbside or close to strong sources, as well as in areas representative of the exposure of the general public, i.e. the urban background. This is likely to require a good number of ambient measurement sites in the urban area which can be used for validation.
- It is anticipated that population level calculations will be carried out after the air quality modelling, which implies that reasonably high-resolution gridded concentrations will be required for the area under investigation. The spatial resolution of the modelling should therefore be in the order of 5m x 5m for the whole city. This level of detail will allow for estimating concentrations at individual receptors whilst also providing for gridded city-wide estimates.
- The modelling system should also be capable of providing results at a sufficiently high number of receptor points to enable subsequent analysis. In regard to the positioning of

the receptor points in the model, pollutants are usually monitored at the height of between 1.5 m and 4 m. The breathing zone for people is normally considered to be around 1.5 m so the model should make predictions at this height as a minimum.

3.4.3 Modelling validation

When comparing dispersion model outputs with measured values there are many evaluation statistics and no single one describes all aspects we may be interested in. We normally use several metrics which in combination elaborate model performance. The performance statistics suggested for use are listed below.

In the definitions below Oi represents the ith observed value and Mi represents the ith modelled value for a total of n observations.

Fraction of predictions within a factor of two, FAC2 (ideal value=1)

The fraction of modelled values within a factor of two of the observed values are the fraction of model predictions that satisfy:

$$0.5 \leq \frac{M_i}{O_i} \leq 2.0$$

Mean bias, MB (ideal value=0)

The mean bias provides a good indication of the mean over or underestimate of predictions. Mean bias in the same units as the quantities being considered.

$$MB = \frac{1}{n}\sum_{i=1}^{N} M_i - O_i$$

Mean gross error, MGE (ideal value=0)

The mean gross error provides a good indication of the mean error regardless of whether it is an over or under estimate. Mean gross error is in the same units as the quantities being considered.

$$MGE = \frac{1}{n}\sum_{i=1}^{N} |M_i - O_i|$$

Root mean squared error, RMSE (ideal value=0)

The RMSE is a commonly used statistic that provides a good overall measure of how close modelled values are to predicted values. In the UK it is recommended that the final model RMSE should be around 10% of the value of the standard being assessed (e.g. 4 µg/m3 when assessing against the 40 µg/m3 NO2 annual mean standard)

$$RMSE = \sqrt{\frac{\sum_{i=1}^{n}(M_i - O_i)^2}{n}}$$

Correlation coefficient, r (ideal value=1)

The (Pearson) correlation coefficient is a measure of the strength of the linear relationship between two variables. If there is perfect linear relationship with positive slope between the two variables, r = 1. If there is a perfect linear relationship with negative slope between the two variables r = −1. A correlation coefficient of 0 means that there is no linear relationship between the variables

$$r = \frac{1}{n-1}\sum_{i=1}^{n}\left(\frac{M_i - \bar{M}}{\sigma_M}\right)\left(\frac{O_i - \bar{O}}{\sigma_O}\right)$$

An important step in any modelling study is comparison of the models with measured values using some or all of the above metrics. The UK's Department for Environment, Food and Rural Affairs (Defra) technical guidance [7] for air quality modelling under the Local Air Quality Management Framework provides a practical workflow for using statistical metrics to investigate model error in a consistent and systematic manner. Figure 3-1 and Figure 3-2 are reproduced from the UK guidance to show how these methods can be integrated into the modelling workflow.

Figure 3-1 UK technical guidance extract 1- comparing model predictions with measurements

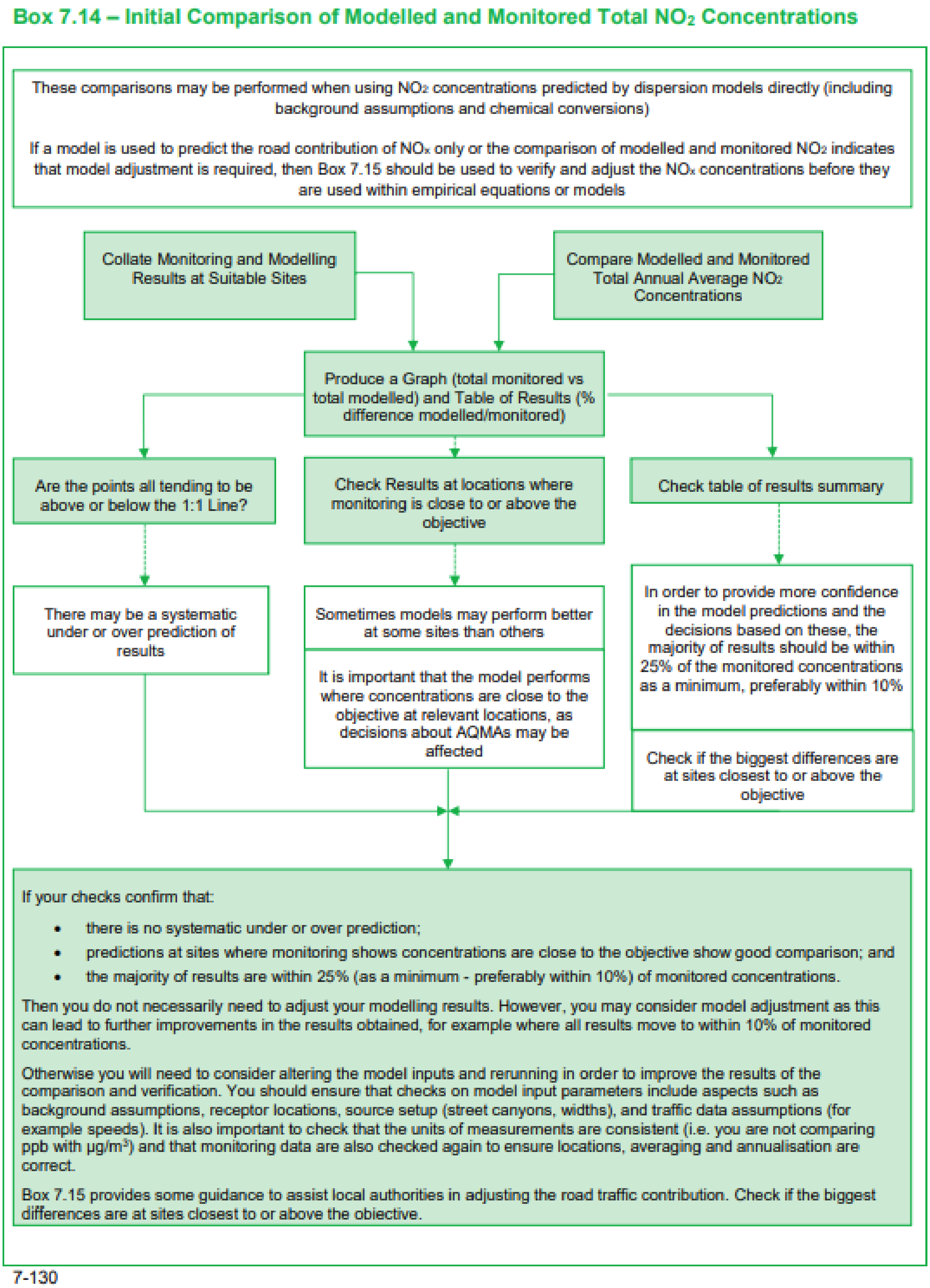
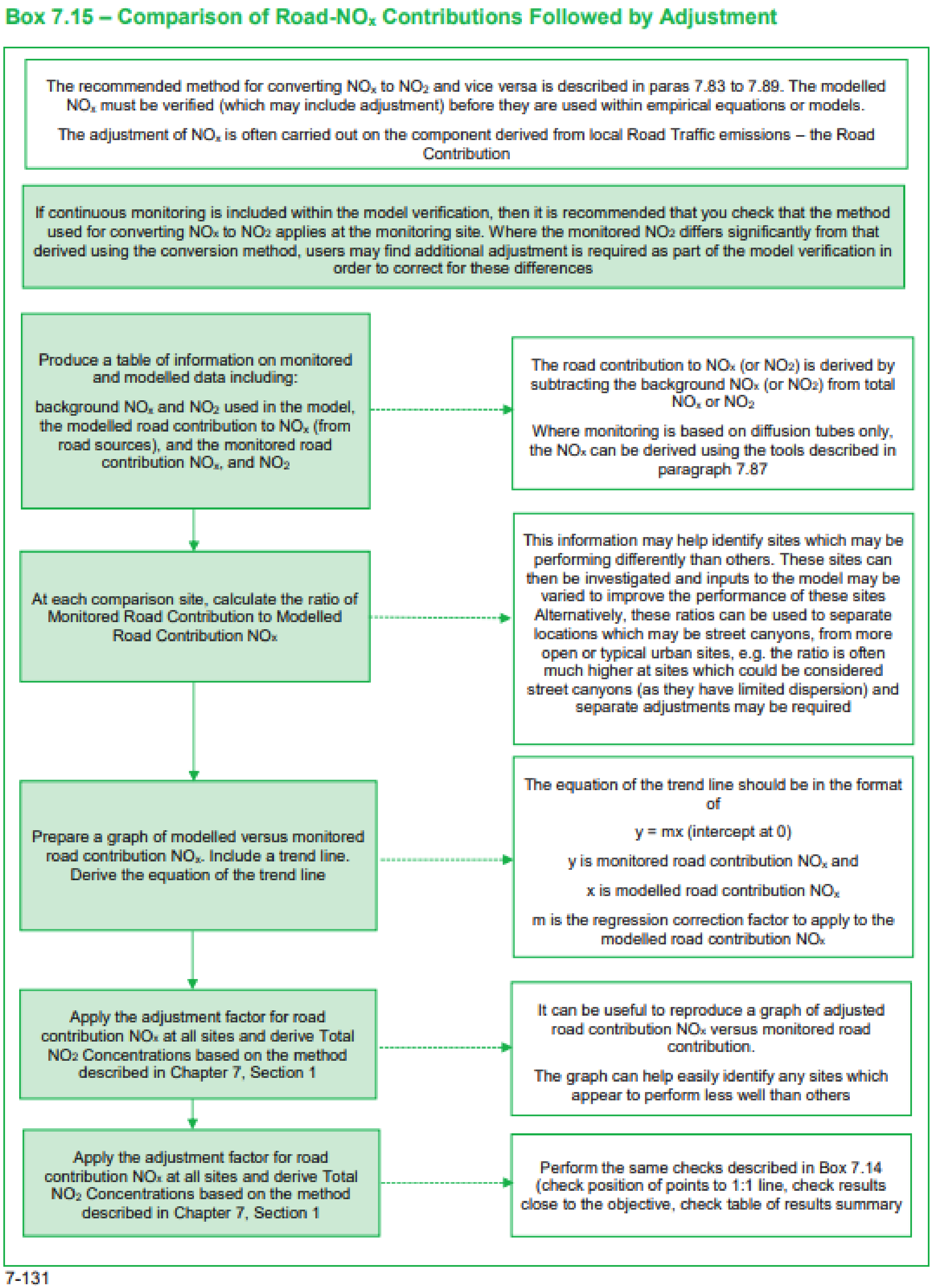


Figure 3-2 UK technical guidance extract 2- comparing model predictions with measurements



3.4.4 Traffic data for modelling emissions to air

This section of the report discusses the types of data that are typically used in emissions and air quality studies when road traffic is the source of most interest. This is of particular interest as road traffic is typically the main source of air pollution in cities. We have reviewed the 'data landscape' in Europe and used our findings to inform the candidate methodologies for emissions and air quality modelling.

3.4.4.1 General considerations

The aim of collating road traffic data for use in an urban scale air quality model application is to create a geospatial road traffic emissions inventory at an appropriate resolution, which then feeds a local or city scale atmospheric dispersion model to calculate pollutant concentrations at locations where human exposure may be present.

As with any computational modelling process, the quality of the model outputs and the ability to answer specific air quality related policy-based tests/questions are dependent on the quality of the input data.

Vehicle emission rates and dispersion characteristics for road sections in a dispersion model are calculated using a combination of data which can broadly be broken down in to:

- Geospatial information – accurate road alignment, road width, height, surrounding building topography, gradients etc.
- Traffic activity data:
    - Average flow for a relevant averaging period e.g. hourly average or daily average,
    - Average vehicle speed over the averaging period of interest – when modelling in urban environments it is important that the spatial resolution of average speed data is enough to characterise emissions well at locations where congestion regularly occurs e.g. approaching busy intersections.
    - Vehicle fleet breakdown – which can be in variable levels of detail, usually dictated by the overall aim of the modelling assessment and required policy evidence.

Traffic activity data for a city scale air quality modelling application is commonly accessed from a traffic model validated with local traffic observations. Traffic model outputs are then supplemented with other observational data to provide the detail necessary for an effective dispersion model.

In the absence of a traffic model it may be necessary to compile information gathered from traffic count surveys and other data sources. This usually means that the air quality modelling is restricted to locations where survey data is available, or where reasonable assumptions regarding proxy traffic activity data can be made.

Further information regarding collating the various data and potential sources are presented in the following sections of the review.

3.4.4.2 Geospatial considerations

The spatial accuracy and resolution of the road traffic emission inventory is an essential aspect of the modelling process. Maximum pollutant concentrations occur close to the roadside and reduce over distances of a few metres. Modern dispersion models can represent these concentration gradients predicting pollutant concentrations at grid resolutions of 1m or less; the spatial resolution of the road traffic emission data should therefore have equivalent accuracy.

In an ideal world for the air quality modeller, traffic model outputs would be provided in a format that is spatially accurate and instantly compatible with the vehicle emission calculation method. However, preparation of a road traffic emission inventory typically requires combination of various spatial and numerical datasets using a geographical information system (GIS). This can be a challenging process when modelling at the city scale with e.g. thousands of separate road link sources; and should be planned carefully to maximise the efficiencies available when using the computational power of a GIS and subsequent transfer of the activity data into an emission calculator and dispersion model.

The geospatial aspects required are:

- Road alignment.
- Road widths, number of lanes can be used as a suitable proxy.
- Road height – where there are bridges, over/under passes, tunnels etc.
- Road gradients – usually calculated using LIDAR or satellite data.
- Building topography – to model street canyon and re-circulation effects.
- Compatibility with other traffic activity datasets – when used in a dispersion model.

Harmonised spatial data sets covering all EU member states are a requirement of the EU INSPIRE Directive [8]. Road traffic data - falls under the Annex 1 'Transport Network' theme. Where available, this includes spatial datasets representing the road network in each EU member state with various useful metadata such number of lanes, speed limits etc. Spatially accurate road centreline alignment shapefiles are also freely available from Open Transport Map [9] which has been derived from the OpenStreetMap[10] dataset – the dataset can be accessed in GIS shapefile format.

Mapping datasets are also available from official national mapping organisations in all member states. Access to these open data is also possible through consolidated data sources such as the 'Open Maps for Europe' online service[11] that will provide free to use maps from more than 40 European countries; created using information from official, national sources.

Compatibility of spatial datasets is an important consideration when compiling road traffic emissions data. The use of consistent data sets allows for efficient joining of the various data in a GIS.

If average traffic speed or journey information (3.4.4.3) is provided by a third-party supplier in a spatial data format, this may also be provided linked to a common representation of the road centrelines in that locality, provision of this data in a format that is compatible with the mapping being used to represent the road links in the air quality model is essential e.g. TeleAtlas mapping is used by the navigation system manufacturer and traffic statistic service provider TomTom[12] , they can also provide it linked to e.g. the UK Ordnance Survey OSGB highways polyline shapefile dataset.

If using traffic model outputs to compile road traffic emissions for use in a dispersion model; often macro-scale traffic models represent traffic activity as vehicle flows and journey times between key nodes on a road network; the simplified visual representation of which does not represent the actual road layout and therefore lacks the required spatial resolution for an air quality model. This requires an, often challenging, interim geo-processing step where the traffic model can be linked or snapped to an accurate representation of the road network [13].

The spatial resolution of other traffic activity data sources will not usually match the required spatial resolution required to account for local building topography when modelling street canyon/re-circulation dispersion effects, this usually requires further manipulation of the shapefile polylines representing road links to match nearby buildings represented as polygons with a height attribute, such as that from UK Ordinance Survey MasterMap Topography Layer Building Heights (https://www.ordnancesurvey.co.uk/business-government/products/mastermap-building). Open source GIS datasets of building footprints are available from Open Street Map (https://www.openstreetmap.org/about).

Road gradients can be considered when calculating vehicle emissions. It is straightforward to calculate these in a GIS using start and end node heights extracted from LIDAR digital surface model (DMS) datasets, ideally at 1m or less resolution. LIDAR datasets for Europe are available on the European Data Portal [14]; coverage and resolution of the available data is likely to be variable.

3.4.4.3 Traffic activity data and potential sources in Europe

The traffic activity data required to calculate pollutant emissions for any specific road link in an air quality model can be summarised very simply into three categories - average vehicle flow, average speed and vehicle fleet breakdown.

The level of detail required in the data for each of these aspects will be dictated by the policy evidence required. For example, effective appraisal of a low emission zone policy will require a detailed understanding of the age of vehicles in the local fleet, typically gathered using site-specific Automatic Number Plate Recognition (ANPR) surveys. Whereas testing of junction infrastructure improvements to reduce congestion and hence emissions during peak traffic periods will benefit from hourly or better resolution speed and flow data.

3.4.4.3.1 Average vehicle flow

Road traffic emission calculation methods used in Europe [15,16] typically consume annual average daily traffic (24hr Annual average daily traffic (AADT)) traffic flows. In some cases, shorter averaging periods of 1-hour may be used for very detailed air quality modelling assessments where that resolution of traffic activity data is available, usually from a micro-scale traffic model.

Where AADT is used, daily and weekly fluctuations in traffic accounting for weekday peak/off-peak periods and weekends are usually simulated using hourly resolution temporal profiles in the dispersion model. The temporal profiles may comprise of one domain wide profile or multiple route or zone-specific profiles.

A recent review of ten European cities access to city level emission inventory data[17] concluded that '*the main common challenge identified by the cities regarding local emission inventories was the lack of traffic intensity data with sufficiently high spatial resolution that is needed to inform local air quality models. Traffic data at street level are usually not available in cities, and it is often also too expensive for cities to compile it*.'

Traffic activity data for city scale air quality modelling applications in recent years have been accessed from traffic models (typically macro-scale) validated with local traffic observations [18,19,20]. This approach avoids the requirement to deploy multiple automatic traffic counters and the associated collation and reformatting of that traffic count data. Use of a traffic model also means that unique road link identifiers from the traffic model can be used to efficiently cross reference the traffic flow data with the geo-spatial representation of the road network that will be used in the dispersion model.

There are also some examples of city scale low emission zone appraisal type modelling assessments where local traffic count data from multiple sites were used [21,22].

Traffic models are usually created at the city or regional level by transport planners. Early collaboration with transport planners and their integration into the project team for a city scale air quality modelling assessment is therefore essential.

Freely available traffic count observations are available at some locations. The density of these observations will be very city specific therefore an initial review of available data will be required for any city scale modelling project. Many European city and central administrations make this data available through European open data portals[23,24]. Repositories collating these types of open traffic data web resources compiled by traffic practitioners and web solution developers are also available[25].

Table 3-1 Candidate traffic data sources for EU countries

Country	Description of available traffic data	Title and/or Reference
Germany	DEMO macroscopic model is available to model national transport	Modelling road transport emissions in Germany – current day situation and scenarios for 2040[26]
Spain (Barcelona)	Multimodel transport model available for Barcelona (BCN-VML)	A coupled macroscopic traffic and pollutant emission modelling system for Barcelona [27]
Norway (Oslo)	Regional traffic model available for Oslo	Evaluation of traffic control measures in Oslo region and its effect on current air quality policies in Norway [28]
UK	Department for transport publish traffic count information	https://roadtraffic.dft.gov.uk/
Switzerland	Traffic volume for 2020 available	https://www.europeandataportal.eu/data/datasets/vm-uvek-shapefile-2020-bundesamt-fur-raumentwicklung-are?locale=en
Brussels Region (Belgium)	Live data from traffic count points is available for the Brussels region	https://data.mobility.brussels/traffic/api/counts/
Venice (Italy)	Traffic monitoring stations were used to estimate the annual traffic from seasonal counts	Annual average daily traffic estimation from seasonal traffic counts[29]
International roads	The UNECE publish traffic data for main international roads in Europe in 2005, 2010 and 2015	https://unece.maps.arcgis.com/apps/webappviewer/index.html?id=cf22916b3df741368b8f234d4390e90b

3.4.4.3.2 Average vehicle speed

Robust modelling of road traffic emissions requires a reasonably accurate understanding of average vehicle speeds e.g. using observed or measured average speeds covering the entire road network will provide a much better evidence base than modelling with assumed values or simply using the speed limit for that road section as a proxy.

The following sources of average speed data typically used in air quality modelling applications are presented in order of increasing model uncertainty:

GPS Observations

Link specific average speed or journey time calculated from millions of observations from GPS devices in vehicles currently represents the best available source of measured average speed data. This type of data is now readily available to purchase and provides worldwide coverage from GPS navigation and fleet telematics device providers[30,31,32]. It has the advantage that it provides averages across the entire road network so should mean that emissions at locations where traffic congestion occurs regularly will be well characterised in the air quality model. Ideally it should be provided in a format that is compatible with the mapping being used to represent the road links in the air quality model; this is essential for efficient geospatial processing and linking of the traffic activity data in a GIS or equivalent automated system. GPS measurement data has been successfully used in a various traffic and air quality modelling applications in recent years[33].

Traffic model outputs

Typically, traffic models provide prediction of node to node average journey time estimates, which are then converted to average speed based on the journey distance between each node. Nodes tend to be located at road intersections. Caution should be exercised by the air quality modeller as use of traffic model outputs may not have the spatial resolution required to characterise emissions well at locations where congestion usually occurs. For example, a traffic model link spanning a 500m section of road will provide the average speed over the entire length of the road link, whereas there may be significant regular congestion occurring at a shorter section of that link approaching a busy junction.

Assumed speeds

Average vehicle speeds in an air quality model can be assumed based on local speed limits, or road category. When assumed speeds are used it is common to apply a 'free flow' speed and a separate 'approaching junction' speed to represent slower moving sections of the road network. This approach is uncertain and is likely to be most appropriate in the absence of other more robust observations as described above.

Spot measurements

Average vehicle speed is often measured at automatic traffic count (ATC) sites, or at locations where remote sensing measurements are made. Although useful, spot measurements only provide average speed at that specific location; and will not provide any information about other locations on the road network where congestion or other road infrastructure may affect vehicle speeds. Speed measurement data from ATC sites in European cites may be available from the European open data portals referenced previously. Spot measurements may be able to provide average speeds at the location for individual vehicle types depending on the monitoring equipment used. As with traffic flow data, the density of these observations will likely be very city specific, therefore a review of available data will be required for any project of this type.

Vehicle fleet breakdown

Vehicle fleet breakdown is another aspect of compiling the road traffic emissions inventory which can be in variable levels of detail, usually dictated by the overall aim of the modelling assessment and required policy evidence. It can vary from a very simple split describing the percentage of heavy vs light vehicles to detailed breakdown of vehicle age for each vehicle type.

Vehicle type fleet split

Vehicle type split for individual road links is usually calculated in the traffic model or using automatic traffic count data. As with traffic flow the use of traffic model outputs means that unique road link identifiers from the traffic model can be used to efficiently combine the traffic activity data with the geo-spatial data that will be used to create the road traffic emissions inventory for the dispersion model.

Vehicle age fleet split

The distribution of vehicle age for various vehicle types within the fleet can be either assumed to follow national fleet statistics; or use local observations of the typical fleet age in that locality.

Local observations are typically gathered using automatic number plate recognition (ANPR) surveys at various locations in the city. ANPR surveys also have the advantage of providing additional detail about the vehicle type fleet split that is not available form standard ATC surveys. It may be necessary to deploy these at locations in a city where the fleet is likely to differ based on the local land use e.g. a city centre is likely to have a different vehicle mix than an outer ring road, or roads close to a port will have a higher percentage of Euro VI classification articulated Heavy Goods Vehicles (HGVs).

3.4.5 Tools for modelling road traffic emissions

The air dispersion modelling activities which will be undertaken in cities as part of this program will in turn be reliant on calculations of road traffic emissions. The dispersion model can be considered a straightforward ‘translator’ of the emissions into atmospheric concentrations. Therefore, it is important that the emission model can make the required calculations in the context of overarching considerations, which can be summarised as:

- **Traffic data**- city scale emission calculations are extremely data intensive requiring link level data on traffic volume, speed and fleet mix at a minimum. Intuitively, the requirements of the input traffic data correlate exactly to the inputs of the emission model that will be used (e.g. we can only enter the number of Euro VI rigid HGVs of a certain weight category if their behaviour is already estimated. Alternatively, it may be possible to infer this from general measurements of HGVs that are subsequently scaled using a pre-determined vehicle split that is appropriate for the location). It follows that using more complex emission models makes it very difficult to find the necessary traffic data to operate them, especially at city scale. This tends to mean that simpler traffic emission models are more widely used at scale.
- **Estimating traffic emissions at city scale** can typically only be facilitated using the outputs of a pre-existing road traffic model, which is how most modern cities design, operate and manage their road infrastructure. Hence, traffic models are normally available in the transportation department of any city authority. An additional benefit of using traffic models is they are also used to forecast traffic conditions, so can be used in air quality studies looking at future years (provided there is a forecast of fleet technology change in the locality as this tends not to be included in traffic models which focus on road capacity).
- **Traffic measurements**- of course, the role of traffic measurements is important but there are major hurdles to overcome if working at city scale and relying only on measured data. Measurements of traffic activity are usually only taken in sparse locations through a city and each one represents a highly localised snapshot of traffic conditions at that location. Remote sensing devices provide the opportunity for traffic to be measured at a larger number of sites than conventional traffic measurements, however in order to obtain representative traffic data for a road the measurements would need be taken over a prolonged period. Usually no meaningful assumptions can be derived from one set of measurements of traffic on a given road that can be applied to other roads in a city. If automatic number plate recognition data is available from the traffic measurements this could, providing locations are a made over a spatially representative area with a large number of recorded vehicles, be used to provide some inputs to the emissions model. For example, these measurements could be used provide proportion of cars that are fuelled by petrol, diesel or electric. Usually it is best to assume that the traffic measurements have been used to validate the traffic model, which is conceptually similar to the role of ambient air quality measurements in validation dispersion model predictions. This further allows for calculating future conditions which may be impossible when relying solely on measurements.

The choice of emission model is not proscribed here but we set out some background technical information that can inform the choice, before providing some candidate systems that could be used.

Firstly, we set out the different types of road traffic emission model that we understand to be in current use in European cities:

- **Average speed based**- these partition emissions into hot exhaust, cold start and non-exhaust based on simple curves that describe emissions versus vehicle speed. These assume that every vehicle of a given technology (say, Euro VI diesel car) has the same emission rate for a given speed, regardless of the work carried out by the engine at that speed. This means that phenomena such as stop/start, idling, acceleration and deceleration are either not possible to integrate, or involve the use of proxy methods. For example, using low speeds to represent congestion, or separating a ‘free flowing’ geometrical representation of a link from an additional link with the same geometry, but with superimposed emissions arising from congestion.

Such models usually have a complete representation of fleet technologies (e.g. all Euro standards for all common vehicles) which is localised for the study area through the introduction of local fleet ‘mix’ which in turn weights the calculated emission by the incidence of the various vehicles that comprise the local fleet.

By far the most widely used of these is the COPERT system [15], which is in routine use in many European countries. Another such system is the EMFAC model which is the official emission model of the California Air Research Board. Although the basis of EMFAC is like COPERT it is very coupled to the USA vehicle categorisations making it difficult to deploy in Europe.

- **Traffic situation based**- the simpler emission models can be thought of as representing a "driving pattern", which in turn represents a typical driving behaviour and can be described with the help of kinematic parameters (typically average speed, dynamics). However, emission models are available that calculate emission rates for different situations on different types of roads. The term "Traffic Situation" refers to emissions calculations that attribute driving patterns to different traffic situations (based on statistical analyses). The term "traffic situation" is more flexible than e.g. a static term such as "road section" because different driving patterns occur on the same road (e.g. stop and go at peak times, fast driving during off-peak periods). For example, if the link average speed is 50 kph and it is an urban road, the model can use a default drive cycle that includes a high proportion of acceleration, deceleration, and idle activity as would be expected on an urban road with frequent stops. If the average speed is 100 kph and it is a rural highway, the model may use a default drive cycle that assumes a higher proportion of cruise activity, smaller proportions of acceleration and deceleration activity, and little or no idle activity.

  An example of this type is the Handbook Emission Factors for Road Transport (HBEFA) [34] which is used in some, but not the majority of, European countries. The USEPA Motor Vehicle Emission Simulator (MOVES) model is another example (though this uses “Op-Mode” rather than traffic situation), but again this is highly coupled to the US context by its incorporation of State specific data such fuel use patterns, meteorology, vehicle maintenance rates and fleet technology. This makes it difficult to use in the European context.

- **Vehicle trip based**- the most complex road traffic emission models make their calculations based on individual vehicle trajectories through a domain. This requires sophisticated microsimulation techniques for road traffic modelling which calculates vehicle movements individually through an area, taking account of junctions and other obstacles to flow. These models calculate the trip parameters of millions of individual vehicles through an area so are computationally intensive for a typical city. These vehicle trajectories can then be used to estimate vehicle specific power which in turn can be used to calculate emissions. Such models are extremely resource intensive and not very amenable to use at city scale- they tend to be used on much more localised areas.

The MOVES model described previously can interpret output from traffic simulation models in the form of second-by-second individual vehicle trajectories. These vehicle trajectories for each road segment can be input into the emissions model and defined as unique links- but this produces a very large number of links to process. There are generally no limits on how many links can be defined in this way; however, model run times increase as the user defines more links. A representative sampling of vehicles can be used to model higher volume segments by adjusting the resulting sum of emissions to account for the higher traffic volume. For example, if a sampling of 5,000 vehicles (5,000 links) was used to represent the driving patterns of 150,000 vehicles, then the sum of emissions would be adjusted by a factor of 30 to account for the higher traffic volume (i.e., 150,000 vehicles/5,000 vehicles). Since the vehicle trajectories include idling, acceleration, deceleration, and cruise, separate roadway links do not have to be explicitly defined to show changes in driving patterns. The sum of emissions from each vehicle trajectory represents the total emission contribution of a given road segment.

A compromise can be for the traffic modelling agency to use a trip based traffic model but then aggregate the outputs so that a simpler emission model can then be used to estimate emissions. In our experience this is the most typical workflow where microsimulation traffic modelling has been undertaken in a city.

3.4.5.1 COPERT

The COPERT emissions model (https://www.emisia.com/utilities/copert/) is part of the European Monitoring and Evaluation Programme/European Environment Agency air pollutant emission inventory guidebook and the methodology is consistent with the Intergovernmental Panel on Climate Change Guidelines for the calculation of greenhouse gas emissions. This makes COPERT the most commonly used emission model in Europe.

COPERT contains emissions factors for more than 450 vehicle types and includes emissions from engine operation, cold start emissions and non-exhaust emissions. Information on vehicle km, vehicle technology and average travel speed are used in combination with description of the area being modelled (the choice of urban, rural or highway driving is use to estimate the likely driving situation e.g. idling) and the emissions factors to calculate total emissions for individual road links or a full inventory. The emissions factors are published for each vehicle classification allowing speed-emissions curves to be produced – the curves produced can be updated e.g. with local emissions measurement data such as that from remote sensing with relative ease.

3.4.5.2 HBEFA

HBEFA, Handbook Emission Factors for Road Transport (https://www.hbefa.net/e/index.html), provides emissions factors for vehicle category and technology for six European countries: Germany, Austria, Switzerland, Sweden, Norway and France. In order to use the HBEFA model elsewhere, the country being modelled would need to be compared to the six available HBEFA countries to establish which is most similar.

The emissions factors are derived from real-world measurements and the PHEM vehicle emissions model. The PHEM emissions model is run using real-world fuel consumption information and 365 different driving situations e.g. road type, speed limit, and a weighted average emission factor calculated. Emissions are calculated taking into consideration hot emissions factors, cold start emissions and evaporation emissions, per e.g. vehicle category or emissions concept. Fleet projection information is also available up to 2050 for five of the six countries (Norway projections only extend to 2035).

3.4.5.3 EMFAC

EMFAC (EMission FACtor, https://arb.ca.gov/emfac/) is the emissions models available from California Air Resources Board to estimate emissions from on road mobile sources. The EMFAC model is approved for use by the USEPA for State Implementation Planning and transport conformity analysis. The fleet database is updated using census block group level information on vehicle registrations in the State. The fleet is available from 2001, with projections to 2050. As such, this means the EMFAC model is tailored to the State of California.

Users can provide custom activity and fleet to calculate emissions for a e.g. project or region. Emissions totals are calculated taking into consideration running, idling, start exhaust, diurnal evaporative, hot soak, running loss evaporative, brake and tyre wear emissions.

3.4.5.4 MOVES

The USEPA has its own on road emissions model MOVES (MOtor Vehicle Emission Simulator, https://www.epa.gov/moves), which is the official emissions model for State Implementation Planning outside of California. The information contained within the model (e.g. activity patterns, fuel parameters, implemented rules such as the Safer Affordable Fuel-Efficient Vehicles Rule) is all therefore specific to the USA.

The user is required to provide fleet (age of vehicle), link-based activity data e.g. traffic volume, and information about population and fuel characteristics (the MOVES model provides some default data which can be updated by the user if more recent or detailed information is available). Using the operating mode selected by the user, estimates of acceleration, cruising and average speed are used to calculate emissions from the user inputs for exhaust, evaporative, brake and tyre wear and provide a total emission rate.

3.4.6 Customisation options for the traffic emission model

In addition to the 'type' of emission model that will be deployed, there is the additional question of whether the emission model can accommodate data from the remote sensing campaigns being undertaken in parallel. This is not straightforward to achieve and there is no standard method, especially with the model complex emission models. We do not consider that the remote sensing can be integrated into any of the emission models except for the average speed based systems such as COPERT, and even then, with some important limitations.

As we have discussed, traffic emissions are modulated by a range of factors such as speed, vehicle type and fleet mix. Taking speed as an example, the models can calculate the emissions associated with the full range of speeds, typically ranging from 5 kph to around 130 kph. COPERT for example has specific emission curves for hundreds of individual vehicles across the entire speed range. Hence, it follows that the integration of remote sensing data must have the aim of either:

7. **entirely replacing the emission curves** in the emission model with solely measured data. This implies that ALL emission curves would be replaced, otherwise individual vehicle categories could be prejudiced in the subsequent emission modelling (e.g. if Euro VI diesel cars are based on measured values, but Euro VI buses are not, how can the two be reliably compared).

8. **providing a means to scale the emission curves** in the emission model by sampling enough vehicles in the same categories and speeds. Assumptions could then be made as to whether to use the scaling factors for vehicles not directly measured, or for vehicles with a small sample size in the campaign (e.g. we may have a good sample size for Euro 6 diesel cars greater than 2.0L engine capacity, and might assume that any scaling factors we derive for these can be applied to smaller Euro 6 diesel cars). Further assumptions would be required to deal with the fact that remote sensing does not typically have good sample coverage of the entire speed range. In such cases we could assume that the scaling required for one speed

maps to the whole speed range (e.g. if a Euro 6 diesel car is measured and we find a divergence from the emission model of 150% at a certain speed, it may be justifiable to scale emission rates at all speeds for that vehicle by 150%). An enhancement of this approach might be to use binned speed values to develop more speed specific scaling factors.

The use of remote sensing devices, as conceived in the NEMO project, provides the opportunity to make real-world emissions measurements at many locations across a city given the portable configuration of the sensors. However, in order to entirely replace an emissions curve with solely measurement data would require measurements to be made at a large number of locations for all vehicle categories, including recording enough vehicles to produce a representative emission for each category. It does not seem sensible to expect that replacing the emission curves is achievable within this project as a very extensive remote sensing program would be required that covers all vehicle types operating in all European countries. It is however possible to use the remote sensing data to compare the modelled emission rates for individual vehicle types with the measured values and make updates to the emission curves.

In our methodological section we provide a simple means of making scaling based adjustments to outputs from the COPERT emissions model. It is thought that this offers a pragmatic approach which integrates the remote sensing in such a way as to mitigate any significant biases (e.g. the emission model underestimates diesel car emissions by a factor of 2) that may exist in the emission model. An attractive aspect of the method is that it allows for the integration of a growing measurement database which could gradually improve the emission model over time. Emission curves could eventually be replaced entirely but this would require the remote sensing to be continuous and operated at massive geographical scale.

The list of potential emission models provided above should not be considered exhaustive and there are likely other systems which could be deployed whilst offering a similar balance of functionality and operational efficiency.

3.5 Existing methodology approaches assessing and valuing health impacts

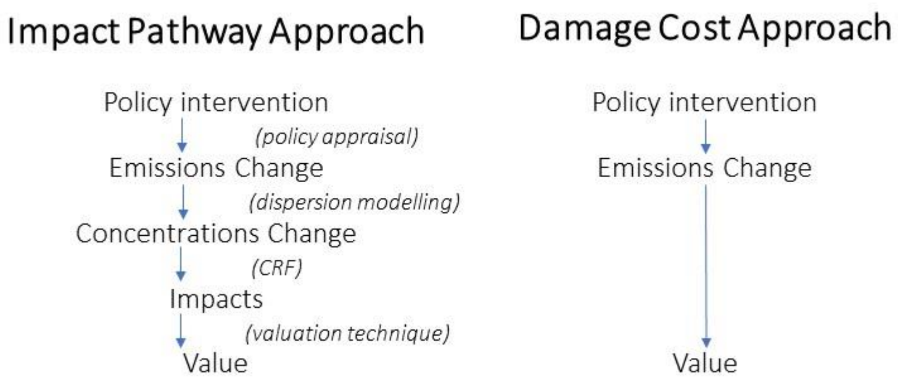
The monetization of the impacts associated with changes in air quality provide key values for any policy design. Methods to assess and monetise the impacts focus on the quantification of the impact that changes in emissions have on human health, environment, and economic activity.

Two different but linked approaches exist to appraise these impacts. Which is selected for use is determined by data and resources available:

Damage Cost Approach: This approach values all damage experienced by individuals as a result of the existence of an externality (e.g. health impacts due to traffic noise) and expresses these impacts as a simple monetised estimate per unit of change. For air pollution, this is a cost per tonne of emission.

Impact Pathway Approach (IPA): This methodology is recommended for use where changes to air quality are the principle objective of the policy or project (which NEMO is trying to achieve). It involves a detailed analysis of the long chain of events preceding the final impact on the exposed population, and therefore relies on a greater level of data and resource inputs.

Figure 3-3: Comparison of Impact Pathway and damage cost approaches



Note damage costs and the IPA are linked. Namely, one would follow the IPA to derive damage costs, which present average impacts per tonne of pollutant emitted. Typically damage costs are derived using national average or source specific values for the different steps along the IPA – e.g. dispersion of pollutants and exposure.

Where estimated changes in air pollutant concentrations are available, the IPA is typically the preferred cost approach as this allows for more precise externality calculations. The IPA approach is applied widely for EC decision-making[1]. It is a simple, logical and sequential description of the evolution of impact following release of a pollutant and can integrate the latest scientific data. Historically, the IPA has been used most extensively in characterisation of air pollutant damages.

In this case, detailed modelling of changes in air pollutant emissions, concentrations and exposure will be available from the preceding modelling steps under the scope of this methodology. As such the IPA can be deployed in this approach. The monetisation of health impacts associated with exposure to air pollution will take changes in air quality appraised under the air quality modelling steps above. It will then need to combine this with other steps on the Impact Pathway Approach to define a monetised impact:

- Population data and overlay of concentrations and population to derive population-weighted average concentrations on a city boundary basis
- Baseline rates of health outcomes on a city boundary basis
- An agreed set of health pathways (and concentration response functions, or CRFs)
- Agreed monetary values per health outcome.

The IPA has been widely deployed by various institutions across varying geographies to appraise air pollutant impacts. However, in each case although the overarching process is the same, the precise methodology has differed, in particular around the set of health pathways included and their CRFs, and also the valuation of health impacts. These variations reflect differences in the interpretation of the underlying epidemiological data and reflect contextual guidance around appraisal methodologies.

The following sections set out and compare varying approaches to appraising air pollutant impacts, to inform our selection of a recommended method in this case.

[1] It is the generally favoured approach, at least in terms of informing, directing and supporting EU Commission policy measures.

Concentration response functions (CRFs) for health outcomes

Under the IPA approach, the available concentration levels can be combined with Concentration response functions (CRFs), expressed in µg/m^3, to convert a given change in air pollutant concentrations into health outcomes. CRFs link a change in exposure to a pollutant to its consequent impacts by expressing a change in a health (or non-health) outcome for a given change in pollutant concentrations. As an example: for an RR of 1.046 per 10µg/m^3 for Working Days Loss due to $PM_{2.5}$ exposure.

Results are defined as the ratio of the incidence observed at two different exposure levels. The RR can therefore be interpreted as the increase in percentages in the relative risk in the reported impact due to an increase in exposure levels of µg/m3. Given CRFs are specified as a change in RR to the baseline, one needs to know the existing risk of the health impact pathways.

The following section provides an overview of the different CRFs by source to give the reader the understanding of the similarity in approach. The following sources are presented:

- EU Handbook on external costs (2020)
- WHO HRAPIE (2013)
- UK Defra IGCB (2020)
- EEA Industrial costs of air pollution (2014) - *(updated version expected to be published soon)*
- US BenMAP (2019)

The following tables show the concentration-response functions from the latest EU's Handbook on the external costs of transport:

Table 3-2: Concentration-response functions (source EU's Handbook on the external costs of transport, based on NEEDS [35])

	Pollutant	Risk group (RG)	RGF value	Age Group (AG)	AGF Value	CRF [1/µg/m^3]
primary and secondary organic aerosol < 2.5, i.e. Particle < 2.5µm	$PM_{2.5}$	all	1.000	Total	1	6.51E-04
	$PM_{2.5}$	all	1.000	MIX	1	9.59E-03
	$PM_{2.5}$	all	1.000	Adults 15 to 64 years	0.672	2.07E-02
	$PM_{2.5}$	all	1.000	Adults 18 to 64 years	0.64	5.77E-02
primary and secondary organic aerosol < 10, i.e. Particle < 10µm	PM_{10}	infants	0.002	Total	0.009	4.00E-03
	PM_{10}	all	1.000	Adults 27 and above	0.7	2.65E-05
	PM_{10}	all	1.000	Total	1	7.03E-06
	PM_{10}	all	1.000	Total	1	4.34E-06
	PM_{10}	Children meeting PEACE criteria - EU average	0.200	Children 5 to 14 years	0.112	1.80E-02
	PM_{10}	asthmatics	0.045	Adults 20 and above	0.798	9.12E-02

	PM_{10}	symptomatic adults	0.300	Adults	0.83	1.30E-01
	PM_{10}	all	1.000	Children 5 to 14 years	0.112	1.86E-01
Ozone [µg/m3] - from SOMO35 The sum of means over 35 ppb (daily maximum 8-hour)	SOMO35	baseline mortality	0.0099	Total (YOLL = 0.75a/case)	1	3.00E-04
	SOMO35	all	1.000	Elderly 65 and above	0.158	1.25E-05
	SOMO35	all	1.000	Adults 18 to 64 years	0.64	1.15E-02
	SOMO35	asthmatics	0.045	Adults 20 and above	0.798	7.30E-02
	SOMO35	all	1.000	Children 5 to 14 years	0.112	1.60E-02
	SOMO35	all	1.000	Children 5 to 14 years	0.112	9.30E-02

The following tables show WHO's HRAPIE[36] project recommendations for concentration–response functions for cost–benefit analysis of particulate matter and nitrogen dioxide (we have excluded ozone as we are not focusing on this pollutant in the methodology):

Whereby:

Group A: pollutant–outcome pairs for which enough data are available to enable reliable quantification of effects;

Group B: pollutant–outcome pairs for which there is more uncertainty about the precision of the data used for quantification of effects.

Table 3-3: CRFs recommended by the HRAPIE project for PM, long-term exposure

Pollutant metric	Health outcome	Group	RR (95% CI) per 10 µg/m3	Source of background health data	Source of CRF
$PM_{2.5}$, annual mean	Mortality, all-cause (natural), age 30+ years	A*	1.062 (1.040–1.083)	European mortality database (MDB) (WHO, 2013a)[37], rates for deaths from all natural causes (International Classification of Diseases, tenth revision (ICD-10) chapters I–XVIII, codes A–R) in each of the 53 countries of the WHO European Region, latest available data	Meta-analysis of 13 cohort studies with results: Hoek et al. (2013)[38]

$PM_{2.5}$, annual mean	Mortality, cerebrovascular disease (includes stroke), ischaemic heart disease, chronic obstructive pulmonary disease (COPD) and trachea, bronchus and lung cancer, age 30+ years	A	Global Burden of Disease (GBD) 2010 study (IHME, 2013)[39], supra-linear exponential decay saturation model (age-specific), linearized by the PM2.5 expected in 2020 under thecurrent legislation scenario	European detailed mortality database (WHO, 2013b)[40], ICD-10 codes cerebrovascular: I60–I63, I65–I67, I69.0–I69.3; ischaemic heart disease: I20– I25; COPD: J40–J44, J47; trachea, bronchus and lung cancer: C33–C34, D02.1– D02.2, D38.1	CRFs used in the GBD 2010 study
PM_{10}, annual mean	Post-neonatal (age 1–12 months) infant mortality, all-cause	B*	1.04 (1.02, 1.07)	European Health for All database (WHO, 2013c)[41]and United Nations projections	Woodruff, Grillo and Schoendorf (1997)[42], based on 4 million infants in the United States
PM_{10}, annual mean	Prevalence of bronchitis in children, age 6–12 (or 6–18) years	B*	1.08 (0.98–1.19)	Mean prevalence from the Pollution and the Young (PATY) study: 18.6% (range 6–41%)	PATY study (Hoek et al., 2012)[43] analysing data from about 40 000 children living in nine countries
PM_{10}, annual mean	Incidence of chronic bronchitis in adults (age 18+ years)	B*	1.117 (1.040–1.189)	Annual incidence 3.9 per 1000 adults based on the Swiss Study on Air Pollution and Lung Disease in Adults (SAPALDIA)	Combination of results from longitudinal studies Loma Linda University Adventist Health and Smog (AHSMOG) and SAPALDIA

Table 3-4: CRFs recommended by the HRAPIE project for PM, short-term exposure

Pollutant metric	Health outcome	Group	RR (95% CI) per 10 µg/m3	Source of background health data	Source of CRF
$PM_{2.5}$, daily mean	Mortality, all-cause, all age	A	1.0123 (1.0045–1.0201)	MDB (WHO, 2013d)[44]	APED metaanalysis of 12 single-city and one multicity studies
$PM_{2.5}$, daily mean	Hospital admissions, cardiovascular diseases (CVDs) (includes stroke), all ages	A*	1.0091 (1.0017–1.0166)	European hospital morbidity database (WHO, 2013e)[45], ICD, ninth revision (ICD-9) codes 390-459; ICD-10 codes I00–I99	APED metaanalysis of four single-city and one multicity studies

$PM_{2.5}$, daily mean	Hospital admissions, respiratory diseases, all ages	A*	1.0190 (0.9982–1.0402)	European hospital morbidity database (WHO, 2013e)[45], ICD-9 codes 460-519; ICD10 codes J00–J99	APED metaanalysis of three single-city studies
$PM_{2.5}$, two-week average, converted to $PM_{2.5}$, annual average	Restricted activity days (RADs), all ages	B**	1.047 (1.042–1.053)	19 RADs per person per year: baseline rate from the Ostro and Rothschild (1989)[46] study	Study of 12 000 adults followed for six years in 49 metropolitan areas of the United States (Ostro, 1987)[47]
$PM_{2.5}$, two-week average, converted to $PM_{2.5}$, annual average	Work days lost, working age population (age 20–65 years)	B*	1.046 (1.039–1.053)	European Health for All database (WHO, 2013)[41]	Study of 12 000 adults followed for six years in 49 metropolitan areas of the United States (Ostro, 1987)[47]
PM_{10}, daily mean	Incidence of asthma symptoms in asthmatic children aged 5–19 years	B*	1.028 (1.006–1.051)	Prevalence of asthma in children based on "severe asthma" in the International Study on Asthma and Allergies in Childhood (ISAAC) (Lai et al., 2009)[48] – western Europe: 4.9%; northern and eastern Europe: 3.5%. Daily incidence of symptoms in this group: 17% (interpolation from several panel studies)	Meta-analysis of 36 panel studies of asthmatic children conducted in 51 populations, including 36 from Europe, (Weinmayr et al., 2010) [49]

Table 3-5: CRFs recommended by the HRAPIE project for NO_2, long-term exposure

Pollutant metric	Health outcome	Group	RR (95% CI) per 10 µg/m3	Source of background health data	Source of CRF
NO_2, annual mean	Mortality, all (natural) causes, age 30+ years	B*	1.055 (1.031–1.080)	MDB (WHO 2013a)[37], rates for deaths from all natural causes (ICD-10 chapters I–XVIII, codes A–R) in each of the 53 WHO Regional Office for Europe countries, latest available data	Meta-analysis of all (11) cohort studies published before January 2013 by Hoek et al. (2013)[38]; RR based on single pollutant models
NO_2, annual mean	Prevalence of bronchitic symptoms in asthmatic children aged 5–14 years	B*	1.021 (0.990–1.060) per 1 µg/m³ change in annual mean NO2	Background rate of asthmatic children, "asthma ever", in Lai et al. (2009)[48] – western Europe: 15.8%, standard deviation (SD) 7.8%; northern and eastern Europe: 5.1%, SD 2.7%, with a recommended alternative of "severe wheeze" in Lai et al. (2009) – western Europe: 4.9%; northern and eastern Europe: 3.5%	Southern California Children's Health Study (McConnell et al., 2003)[51]; coefficient from two-pollutant model with organic carbon (OC) (coefficients from models with PM_{10} or $PM_{2.5}$ are higher)

				Prevalence of bronchitic symptoms among asthmatic children 21.1% to 38.7% (Migliore et al., 2009; McConnell et al., 2003)[50,51]	

Table 3-6: CRFs recommended by the HRAPIE project for NO_2, short-term exposure

Pollutant metric	Health outcome	Group	RR (95% CI) per 10 µg/m3	Source of background health data	Source of CRF
NO_2, daily maximum 1-hour mean	Mortality, all (natural) causes, all ages	A*	1.0027 (1.0016–1.0038	MDB (WHO, 2013d)[44], rates for deaths from all natural causes (ICD-10 chapters I–XVIII, codes A–R) in each of the 53 countries of the WHO European Region, latest available data	Air Pollution and Health: a European Approach (APHEA)-2 project with data from 30 European cities; RR adjusted for PM_1
NO_2, daily maximum 1-hour mean	Hospital admissions, respiratory diseases, all ages	A	1.0015 (0.9992–1.0038)	European hospital morbidity database (WHO, 2013e)[45], ICD-9 codes 460–519; ICD-10 codes J00– J99	APED meta-analysis of four studies published before 2006; coefficient from single-pollutant model WHO (2013f)[52] noted that the estimates for this pollutant–outcome pair were robust to adjustment to co-pollutants
NO_2, 24-hour mean	Hospital admissions, respiratory diseases, all ages	A*	1.0180 (1.0115–1.0245)	European hospital morbidity database (WHO, 2013e)[45], ICD-9 codes 460–519; ICD-10 codes J00– J99	APED meta-analysis of 15 studies published before 2006; coefficient from single-pollutant model WHO (2013f)[52] noted that the estimates for this pollutant–outcome pair were robust to adjustment to co-pollutants

The following tables show UK's DEFRA "Air Quality damage cost update 2019"[53] recommendations for concentration–response functions applied in updated damage costs (% per 10µg/m^3 change in concentration for relevant averaging period):

Table 3-7: CRFs applied in updated damage costs (% per 10µgm^{-3} change in concentration for relevant averaging period)

Pollutant	Pathway	Air pollution metric	CRF type	Reference change in concentration (ugm-3)	Low % or Odds ratio change per 10ugm-3 change in pollutant	Central % or Odds ratio change per 10ugm-3 change in pollutant	High % or Odds ratio change per 10ugm-3 change in pollutant
$PM_{2.5}$	Chronic mortality	Annual Average	Relative Risk (RR)	10	4	6	8
PM_{10}	Respiratory hospital admission	Annual Average	Relative Risk (RR)	10	0.8	0.8	0.8

PM_{10}	Cardiovascular hospital admission	Annual Average	Relative Risk (RR)	10	0.8	0.8	0.8
SO_2	Deaths brought forward	Annual Average	Relative Risk (RR)	10	0.6	0.6	0.6
SO_2	Respiratory hospital admission	Annual Average	Relative Risk (RR)	10	0.5	0.5	0.5
NO_2	Respiratory hospital admission	Annual Average	Relative Risk (RR)	10	0.5	0.5	0.5
NO_2	Chronic mortality	Annual Average	Relative Risk (RR)	10	0.8	2.3	3.7
PM_{10}	Chronic Bronchitis	Annual Average	Relative Risk (RR)	10	1.02	1.32	1.71
$PM_{2.5}$	CHD	Annual Average	Hazard Ratio (HR)	5	1	19	42
$PM_{2.5}$	Stroke	Annual Average	Hazard Ratio (HR)	5	2.1	6.4	10.9
$PM_{2.5}$	Diabetes	Annual Average	Relative Risk (RR)	10	2	10	18
$PM_{2.5}$	Lung cancer	Annual Average	Relative Risk (RR)	10	4	9	14
NO_2	Asthma (Adults)	Annual Average	Odds Ratio (OR)	10	1	10.4	1.08
NO_2	Diabetes	Annual Average	Relative Risk (RR)	10	2	5	7
NO_2	Lung cancer	Annual Average	Relative Risk (RR)	10	0	2	3
$PM_{2.5}$	Asthma (Older Children)	Annual Average	Odds Ratio (OR)	10	1.22	1.48	1.97
NO_2	Asthma (Small Children)	Annual Average	Odds Ratio (OR)	10	1.01	10.8	1.12
NO_2	Asthma (Older Children)	Annual Average	Odds Ratio (OR)	10	1	1.03	1.06

The following tables show EEA's Response function from "Revealing the cost of air pollution from industrial facilities in Europe"[54]:

Whereby:

Population factor 1: This factor accounts for most functions applying to only part of the population. For example, the chronic mortality function (deaths) is applicable only to those aged over 30, who account for 62.8 % of the population in the modelled domain. While the table provides European average figures, the modelling undertaken to generate the results that follow used national data.

Population factor 2: This factor accounts for some functions being expressed per thousand or per hundred thousand of population.

Table 3-8: Incidence Data, Response function and valuation data for quantification of health damages linked to PM exposure 2010 (2005 prices)

Effect – Core Functions	Population Factor 1	Population Factor 2	Incidence Rate	Response Function
Chronic mortality (deaths, VSL valuation)	0.628	1	1.61%	0.60%
Chronic mortality (life years lost, VOLY valuation)	1	1.00E-05	1	65.1
Infant mortality (1-12 months)	0.009	1	0.19%	0.40%
Chronic bronchitis, population aged over 27 years	0.7	1	0.378%	0.70%
Respiratory hospital admissions, all ages	1	1.00E-05	617	0.114%
Cardiac hospital admissions, all ages	1	1.00E-05	723	0.06%
Restricted activity days (RADs) working age population	0.672	1	19	0.475
Respiratory medication use by adults	0.817	0.001	4.5%	90.8
Respiratory medication use by children	0.112	0.001	20%	18
Lower respiratory syndromes (LRS), including cough, among adults with chronic symptoms	0.817	1	0.3	0.130
LRS (including cough) among children	0.112	1	1	0.185

As the above tables on various approaches of CRFs show us, there are lots of similarities across approaches, with slight variations. Since NEMO is a project by the EU Commission, the above chosen sources seem the most relevant. At the time of writing, the EEA study 2021 was not publicly available yet, but shall be part of the guidance throughout the project going forward. In its absence still, the EC handbook appears most appropriate as is a various recent version and also had been developed for EU context.

Baseline epidemiological data - country specific data availability

As noted above, CRFs define the impact of changes in exposure to air pollutant relative to the baseline incidence or prevalence of a given health outcome. As such, for the given city or location, we will need to gather appropriate data regarding existing levels of health outcomes.

A first step would be to gather and review location or country-specific health data. This could include national published statistics such as hospital or GP data. It is preferable to use as location-specific data as possible, but it is likely that such data would more commonly be available at national level.

Where location or country specific data is unavailable, it is typical to fall back on other data sets such as those mentioned in the Table above summarising WHO's HRAPIE CRFs, or the WHO's Global Burden of disease study, which may provide country specific data and /or a recommended baseline for appraisal.

Population at risk

In order to manipulate calculation for health impacts, one needs population data that represent the impact boundary, or an estimate thereof. Since this study is looking at often very localised air quality impacts, the population data must reflect that. The population data needed shall be provided by the Air Quality modelling and the Air Quality domain.

The population-weighted values generated by the Air Quality Modelling are the used in the health concentration-response functions and shall result in the various health impacts.

Calculation of the change in incidence

The estimation of the change in incidence due to a decrease of 1 µg/m^3 of $PM_{2.5}$ or NO_2 is different depending on whether the CRF is based on the Relative Risk, Hazard Risk or Odds Ratio.

Relative and Hazard Risks:

The change in incidence (ΔI_i) per 100,000 inhabitants when the CRF is based on either the Relative Risk or Hazard Risk is estimated as the product between the concentration of the pollutant, the baseline incidence, and the population as in Equation 2:

$$\Delta I_i = \frac{\Delta C_{Pol}}{C_{Inc}} \cdot \frac{RR}{100} \cdot \frac{N}{10^5} \cdot I_i \qquad (2)$$

Where:

ΔC_{Pol} is the concentration of a given pollutant ($PM_{2.5}$, NO_2).

C_{Inc} is the concentration increment on which the CRF is based (5 or 10 µg/m^3).

RR is the Relative Risk (or Hazard Risk, if applicable).

N is the total population of the United Kingdom.

I_i is the age- and gender-weighted incidence of a disease *i*.

Odds Ratio:

The estimation of the change in incidence (ΔI_i) per 100,000 inhabitants when the CRF is based on the Odds Ratio (OR) is more complex, as it requires an estimate the odds of reporting the disease at the new concentration (κ_i) first, as in Equation 3:

$$\kappa_i = \exp\left(-\ln(OR) \cdot \frac{\Delta C_{Pol}}{C_{Inc}} + \ln \frac{I_i}{10^5 - I_i}\right) \qquad (3)$$

The change in incidence (ΔI_i) per 100,000 inhabitants can be then estimated as a function of the odds of reporting the disease at the new concentration (κ_i) as in Equation 4:

$$\Delta I_i = \frac{N(1+\kappa_i)}{\kappa_i(I_i-1)+I_i} \qquad (4)$$

In the case where relative risk values were based on concentration increments of 5 μg/m^3 (C_{inc}), these were used in preference to those extrapolated in the PHE report to a 10 μg/m^3 concentration increment base. This was done in order to be consistent with the methodology explained above, since the extrapolation of relative risk values made in the PHE report was non-linear and the damage cost approach assumes a linear scaling.

Estimation of air pollution impacts and costs

For some impact pathways, depending on how the CRF is expressed, it may be necessary to include an additional calculation step to facilitate the monetisation of effects. Specifically, for morbidity pathways that calculate a change in incidence of a condition, it is typical to convert this into a change in Quality-Adjusted Life Years (QALY) to then monetise these effects.

Quality-Adjusted Life Years (QALY) lost are then multiplied by the value of a life year (VOLY) to obtain the costs. Costs have been calculated for the change in disease incidence for the considered health outcomes. The calculation of QALY loss requires utility weights for the different diseases, which are then multiplied by the change in incidence as in Equation 5:

$$QALY\ Loss_i = (1 - w_i) \cdot \delta_i \cdot \Delta I_i \qquad (5)$$

Where:

$QALY\ Loss_i$ are the quality-adjusted life years for disease *i*.

w_i is the utility weight for disease *i*.

δ_i is the discounted duration of disease *i*.

Utility weights are expressed by a number of sources, e.g. WHO Global Burden of disease work. An example of such weights originally published in Sullivan et al., (2011) [55] for the United Kingdom as presented in the table below (these are used for the calculation of UK Defra damage costs). Males and females were allocated the same EQ-5D score and the diseases were mapped onto conditions listed in the publication using matching, or closest matching ICD-9 Categories. These weights represent the QALY loss associated with each condition whilst living with the condition.

Table 3-9: List of EQ-5D values (QALY weights) allocated to males and females for each disease

Disease	w_i	Mapped ICD-9 Categories
Asthma	0.722	ICD-9 493 Asthma
CHD	0.61	ICD-9 410 Acute Myocardial Infarct
Stroke	0.63	ICD-9 433 Precerebral Occlusion
Diabetes	0.66	ICD-9 250 Diabetes Mellitus
Lung cancer	0.56	ICD-9 162 Malignant Neoplasm Trachea/Lung

The duration of the disease is reflected in the δ_i, which is calculated according to Equation 6:

$$\delta_i = 1 \quad \text{if } D = 1 \qquad (6a)$$

$$\delta_i = 1 + \sum_{j=2}^{D} (1+r)^{1-j} \quad \text{if } D > 1 \qquad (6b)$$

Where:

D is the average years of duration of the disease.

r is the discount rate (*r*=0.04[2]).

The average years of duration of the disease can be calculated using the WHO DISMOD II model [56] and estimated based on the years of life with disability (YLD). For example, the specific average years of duration for the diseases included in the UK Defra damage costs are presented in Table 3-10. As the duration of the disease has been taken into consideration, the QALY loss (which, by definition, looks at the impact of living with the condition for a single year) can provide an indication on the lasting effects that conditions have beyond the first year.

Table 3-10 – Average and discounted duration of disease

Disease	D [years]
CHD	9.50
Asthma in Adults	23.60
Asthma in Children	36.20
Stroke	14.80
Diabetes	9.10
Lung cancer	1.80

By combining the change in incidence, with the QALY weight of living one year with the disease, and the (discounted) duration of the disease, this then calculates the cumulative QALY weight over the expected duration of the diseases associated with all incidences of the disease in a given year.

Finally, the costs produced by increases in the concentration of either $PM_{2.5}$ or NO_2 is the product of the valuation of a QALY loss and the quality-adjusted life years for disease *i* as in Equation 7:

$$Cost_i = QALY\ Value \cdot QALY\ Loss_i \qquad (7)$$

These costs can then feed into the policy modelling.

Valuation

The monetary equivalent of each impact is calculated by simple multiplication[3] of each impact category with a corresponding marginal damage cost factor. This yields the monetary equivalent (damage) of the change in impacts following from a given change in exposure.

This is the typical calculation chain following the impact pathway approach. It can be applied to quantifying and monetizing the impacts of an emission source, a country or region or in our case city boundaries of emission mitigation measures and scenarios.

[2] Consistent with EC Better Regulation Toolbox

[3] This is possible only because the CRFs used here are linear associations with exposure.

3.6 Recommended methodology

3.6.1 Air Quality Modelling

This section sets out the main features of a methodology that can be applied at city scale for assessing the concentrations of criteria air pollutants arising from road traffic. The methodology is not exhaustive as, for example, there will always be a need to adapt to the availability (or otherwise) of data sources in a given city or region.

3.6.1.1 Emissions from road traffic

We suggest that the emissions modelling is carried out using the COPERT emission model. This appears to offer a good balance of functionality, vehicle coverage and customizability whilst being consistent with the traffic activity data landscape as we understand it. It should be noted that COPERT can itself be part of a wider emission modelling system which uses its governing equation and vehicle specific coefficients, with subsequent steps to localize and total the emissions according to the local conditions being handled automatically. If the modeler has the means to make such a tool (or use a model they have already) whilst retaining consistency with COPERT

The methods described in the EMEP/EEA air pollutant emission inventory guidebook[57], specifically the road transport section (1.A.3.b.i-iv) are well suited to this application, especially if the Tier 3 methodology is used[58][59]. The Tier 3 methodology is fully supported by the capabilities of the COPERT emission model, which is provided by the EEA in spreadsheet form[60]. The model was updated in 2020 and this version should be used.

The spreadsheet model will require additional effort to manipulate the outputs in three main areas:

1) **The calculated emissions are representative of a single vehicle** of each of the given technology categories, running at a speed defined by the user, and are provided in grams/kilometer. In reality, each road source in a city will comprise a locally specific mix of vehicle categories which will need to be estimated in each case. The modeler will need to develop their own method for combining the emissions representing the volume, speed and type of vehicles operating in the modelling domain.
2) **The calculated emissions are not representative of locally measured emissions**- a key aspect of this project is the remote sensing campaigns which will measure emissions at source from a wide range of vehicles in different locations. To integrate the remote sensing the modeler should compare the measured emissions with modelled emissions from COPERT. It will be necessary to derive scaling factors or functions that can be applied to bring the model closer to the measurements, most likely by comparing average emissions calculated for a given vehicle category, with the same category in COPERT.
3) **The calculated emissions are not spatially allocated**. The emissions need to be linked to a spatial representation of the road network that will eventually be passed to the chosen dispersion model. This is a straightforward task in any GIS system.

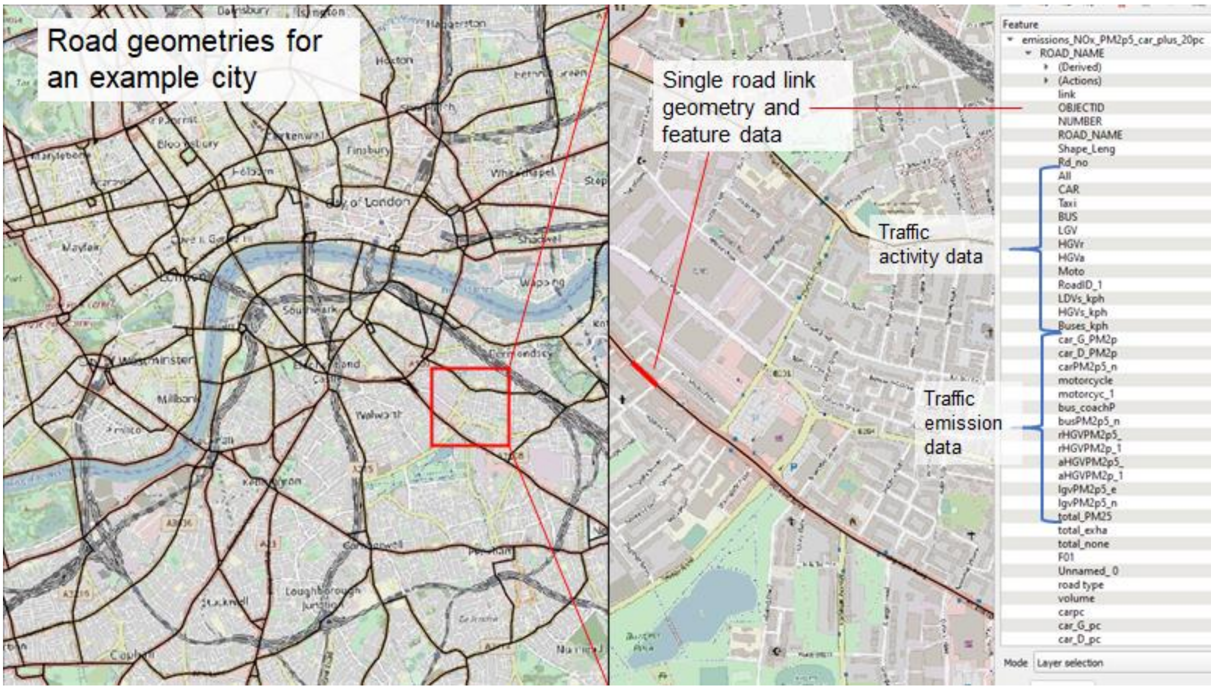
An alternative approach is to take the emission factors, or the coefficients used to calculate them from the spreadsheet and create a custom tool which allows the points above to be addressed. In this case, any such tool should yield the same emission rates as the spreadsheet (this should be straightforward given the coefficients for the emission calculations and the form of the equation is clearly set out in the spreadsheet).

Given these points, the following general steps are suggested for calculating road traffic emissions across a district or city, whilst incorporating remote sensing measurements, and ensuring that the format of the emissions is correct for use in dispersion models.

1) **Obtain geometries for the roads in the study**- most likely in a polyline shapefile format. Each road should be represented by a discrete feature for which traffic activity data can be ascertained. The road geometries need to be spatially accurate so that reliable source/receptor distances can be derived, which will make the dispersion modelling as accurate as possible (this will also help to support model validation against ambient measurements).
2) **Populate the geometries with road traffic activity data**- as a minimum this should include speed, daily vehicle flow, and the split of HDVs, buses, diesel cars, gasoline cars, LGVs and motorcycles. This data will typically be derived from a combination of observations and traffic models.
3) **Run the COPERT model to generate emission factors for all potential vehicles operating in the locality**- emission rates should be calculated at representative speeds for each vehicle type. At this point the emission rates are not locally specific as the vehicle technologies are assumed to emit at the same rate regardless of the country they are in. Alternatively, take the general COPERT equation (which is not vehicle specific) and the relevant coefficients (which are) and derive a project specific emission model that can be shown to produce the same emission rates as the EEA spreadsheet.
4) **Perform a statistical analysis of the relationship between remote sensing measurements and COPERT modelled values**- this can be considered a validation of the emission model that is conceptually similar to the validation that will be undertaken of the modelled concentrations against ambient measurements. The method used for this will largely depend on the nature of the remote sensing measurements, in particular their coverage of the operating speeds for vehicles in COPERT. The most basic method involves simply scaling modelled emissions from each vehicle category by the ratio of measured over modelled emissions- which should be sufficient to account for significant biases in the modelled values (say, a tendency to underestimate emissions from buses of a certain Euro category). Alternatively, for each vehicle category an equation could be derived which scales the emissions from COPERT differently across the speed range. This would require the remote sensing to yield sufficient data points across the speed range available in COPERT.
5) **Produce 'remote sensing scaled' emission factors for each vehicle category**- thus amending the modelled emission rates for each vehicle category before they are combined in to totals for each road link.
6) **Estimate the split of vehicle technologies within each category operating in the area**- these will be used to take the calculated emissions from COPERT (with remote sensing scaling) and weight them according to the prevalence of the different engine technologies and sizes in the area. Countries who prepared national inventories will likely have published splits that can be applied if no locally specific data is available.
7) **Create a combined emission rate for each road feature in the geometry** that represents the weighted value of emissions given the volume and mix of vehicles on that road link. Broadly speaking this will involve taking the volume of traffic, splitting it according to the volume and mix of vehicles, and separately calculating the remote sensing scaled emission for each category. The emissions are then recombined into a single value per road feature (in grams per kilometer, and perhaps grams/kilometer/second), which is passed to the air quality model. As well as the combined emission rate, it can be useful to retain split emissions from high level vehicle categories to use in source apportionment (e.g. combine emissions into buses, HDVs, cars, LGVs, motorcycles). The result should be a geospatial dataset resembling the example in Figure 3-4.
8) **Calculate emission mass from the road network for use in damage cost calculations**- this will involve multiplying the combined emissions in each road feature by 365 to yield an annual

value- presuming the traffic activity represents a daily flow. This value will be in grams, so should be converted into tons for compatibility with the damage cost estimates.

Figure 3-4 Example of road geometry representation with traffic and emissions data



In summary, the output data products from the emission modelling which are required for dispersion modelling and economic analysis are:

1) **Remote sensing adjusted and geospatially allocated emission rates** for each road link comprising daily emissions from the representative vehicle fleet mix and volume. Emission rates for the dispersion modelling should be in units of grams/kilometer or grams/kilometer/second.
2) **Emission inventory based on the same inputs, calculated at the link level** with data in units of tons per year for the main pollutants (NOx, PM, perhaps CO_2 if it has to be included in economic modelling).

3.6.1.2 Dispersion modelling

There are a wide range of air quality models that may be used for the assessment of concentrations, many with similar capabilities that we described earlier. Therefore, it is less imperative to suggest an air quality model over any others- suffice to say the model should be capable of addressing the points we discussed earlier.

By this point, the assumption is that there is a detailed, spatially allocated road traffic emissions inventory which will be one of the primary inputs to the dispersion model. The spatial accuracy of the concentrations is entirely dependent on the spatial accuracy of the emissions model.

The workflow implied in city scale road traffic pollution modelling is very complex, so it is not possible to be prescriptive on every detail. However, we can provide some guiding principles that should help in designing and executing such studies. Hence, the following general steps are suggested for calculating road traffic concentrations across a district or city, so that the outputs may be used in economic and health impact assessment.

1) **Select a suitable dispersion model**- a list of candidate systems is provided in an earlier section. The system should be fit for purpose, especially in its ability to represent hundreds, or even thousands of discrete road sources in a spatially detailed and accurate manner. The model must

be able to accommodate either line, line area, or volume sources, which are the most representation of road sources in dispersion models.

2) **Process the geospatially allocated emission inventory into 'model ready' format**- some dispersion models can read the inventory in its native format, e.g. Esri shapefile. Others will require some processing to input the spatial inventory into the model- for example in some models this is done by providing the vertices of the line sources with the appropriate emission rates. Each source should have its own unique identifier so that the data can be traced back through the emission calculations. Care should be taken that the mass of the emissions presented to the dispersion model matches the emission inventory- i.e. any conversions are not 'lossy'.
3) **Set a representative receptor grid covering the domain**- pollution gradients around road sources are sometimes very pronounced so it is important to use a high-resolution receptor grid in the model. Depending on the model this can come with a large computational overhead when executing the simulations. In city environments it is recommended that the receptor grid should be set to a maximum of 5m. For a 10km x 10km domain this represents 4 million receptor points. Alternatively, a lower resolution may be used with appropriate interpolation carried out post-process- though this comes with the risk of missing important features in the concentration distributions. The model domain could also be split across several model runs if necessary.
4) **Obtain background concentrations of the pollutant of interest**- the road dispersion model requires background concentrations of the pollutant of interest which accounts for non-road sources of pollution. This can be derived in a number of ways- most commonly from a regional scale air quality model or background measurements. In the case of measurements, these can be interpolated to provide a continuous background pollution map for a city- though care should be taken than the road traffic being modelled is not double counted because it also contributes to the measured background values. If regional modelled values are being used, these tend to be quite low resolution- typically in the order or 1km to 10km resolution. Using these data 'as is' can introduce quite significant edge effects into the total concentrations. This can be avoided by resampling the background model grid to a higher resolution in combination with a smoothing routine (e. g. a by passing a 2D filter over the grid based on a moving average, nearest neighbors, inverse distance to a power- all of which are available in GIS systems). The background data should be formatted so as to be compatible with the dispersion model being used. This can involve either deriving background values at the same receptor coordinates or by saving it as a georeferenced raster which can be added to the road model values post process.
5) **Obtain ambient measurements of the air pollutants under consideration**- for the purposes of economic and health assessment it is most common to use annual average ambient measurements of air pollutants. The availability of good quality ambient measurements is fundamental to the success of the modelling exercise. The data should have good spatial coverage and represent a mix of urban, suburban and background sites. The site metadata should be recorded- in particular, the sampling height and inlet coordinates (in a projected coordinate reference system, e.g. UTM) are required as they will be input to the dispersion model as discrete receptors during validation. The precise location is required (accurate to plus or minus 2m or so) so that the distance to the nearest sources is accurate, which helps when validating the results.
6) **Obtain meteorological data for the model**- each model has its own exact requirements but in general they all rely on wind speed and direction and proxy data which is used to derive convective turbulence in the atmospheric boundary layer (e.g. cloud cover). The data should be

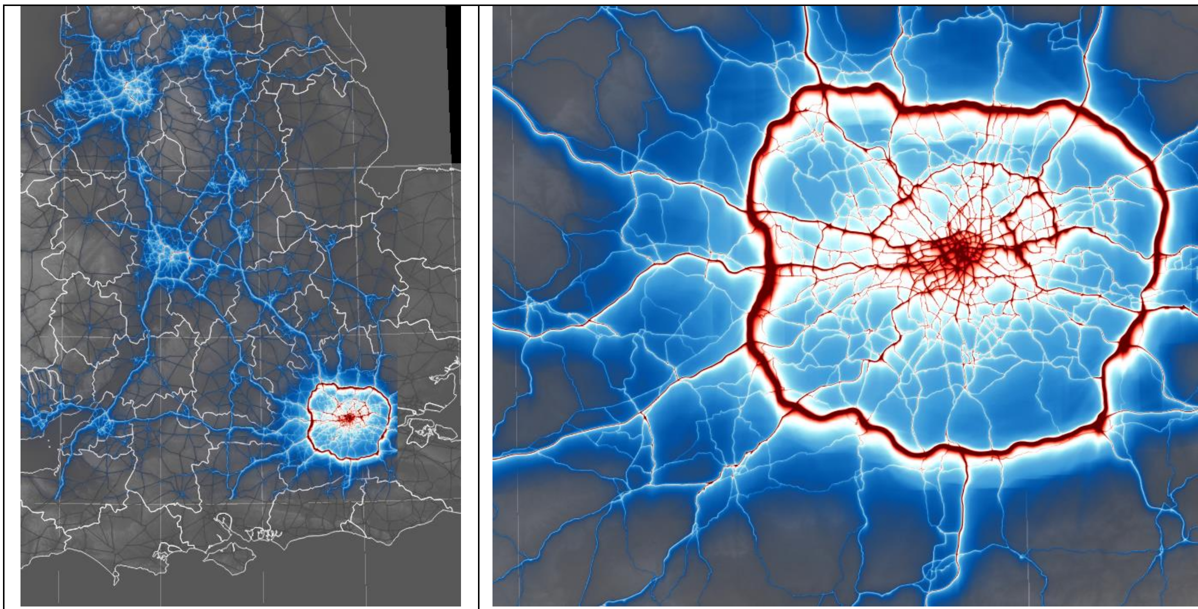
derived from at least one met station, which should be operated to World Meteorological Organization [61] (or relevant domestic) standards- commonly these are situated at airports though most countries operate an extensive network of stations through their national meteorological agency. Most models have their own meteorological processors which derive a range of parameters from the quite simple input meteorology. For example, the wind speed/direction can be processed into friction velocity, mechanical turbulence, and mechanical boundary layer height. The cloud cover data is processed into solar radiation, convective velocity scale and convective boundary layer height. In most cases, road traffic-based dispersion model results are quite insensitive to convective processes and are more sensitive to mechanically driven processes- hence it is natural to focus on the quality of the wind data presented to the model. The USEPA provide guidance62 on the topic of acquiring, processing and quality assuring meteorological data for dispersion models. The guidance also provides data capture standards, and methods that can be used for gap filling missing data where necessary.

7) **Set the dispersion model appropriately for the situation**- several parameters need to be set in the model, mainly for the purposes of representing urban conditions. Most commonly these are surface roughness length, surface albedo and Bowen ratio- default values are available in technical guidance [61,63]. The model should be set to provide annual mean concentration outputs, at the receptor grid defined earlier.
8) **Run the simulation and process the outputs**- running the dispersion model can be computationally expensive so it can be useful to test the simulation on a smaller subset of the meteorological data to check for errors. When any problems are addressed the simulation can be carried out for the whole calendar year- which should be set to the same year as the ambient measurements and meteorological data. At this stage the background concentrations can be added and any post-processing to account for chemical transformations can be done. In the case of NOx/NO_2 it is common to use empirical formulations to convert from modelled NOx concentrations to NO_2.
9) **Validate the simulation results at measurement sites and make any corrections to the outputs**- the measured and modelled values of the pollutants under consideration should be compared (suggested metrics for the comparison are provided earlier). At this stage the modeler may adjust the model results to account for under or over prediction that is found during the validation (e.g. based on the slope coefficient from a linear regression). A simple correction factor can be derived, based on the measurement sites, which can then be applied to all other receptor locations. This simple approach implies that the model diverges from measurements in a linear manner, which may not be the case. In such cases it may be better to undertake a polynomial regression and use the derived expression to correct the model. In some cases, if there is enough measurement data, a zonal approach can be taken to model validation. This can be done by separating out locations that are highly urbanized from those that are not, developing separate correction factors and applying them to locations in the receptor grid that have the same characteristics. It is recommended that the validation is focused on the annual mean concentrations as these are the main input to any analysis which follows (economic or health studies). It should be noted that simply correcting the model results based on the first pass validation results is not a good idea- the modeler should iteratively investigate potential causes of any divergence and correct them prior to accepting the final results.
10) **Prepare mapped concentration fields and numerical results at key receptors**- the output concentrations should be processed into a format that allows mapping of the outputs at the city scale. This will likely involve post-processing in a GIS to a format such as GeoTIFF. In addition, numerical results should be extracted at key locations, should a comparison with air quality

standards be required. An example of a GeoTIFF with concentration data from a city-level air quality model is shown in

11) **Derive ancillary data from the results**- the modeler may wish to derive data such as population weighted exposure, neighborhood average concentrations, maximum or minimum values and a range of other statistical representations of the outputs. For health studies it is likely that population density data would be overlayed on the concentration fields to make the calculation in a GIS. For neighborhood averages a set of boundary polygons would be used and the values inside each computed in the GIS.

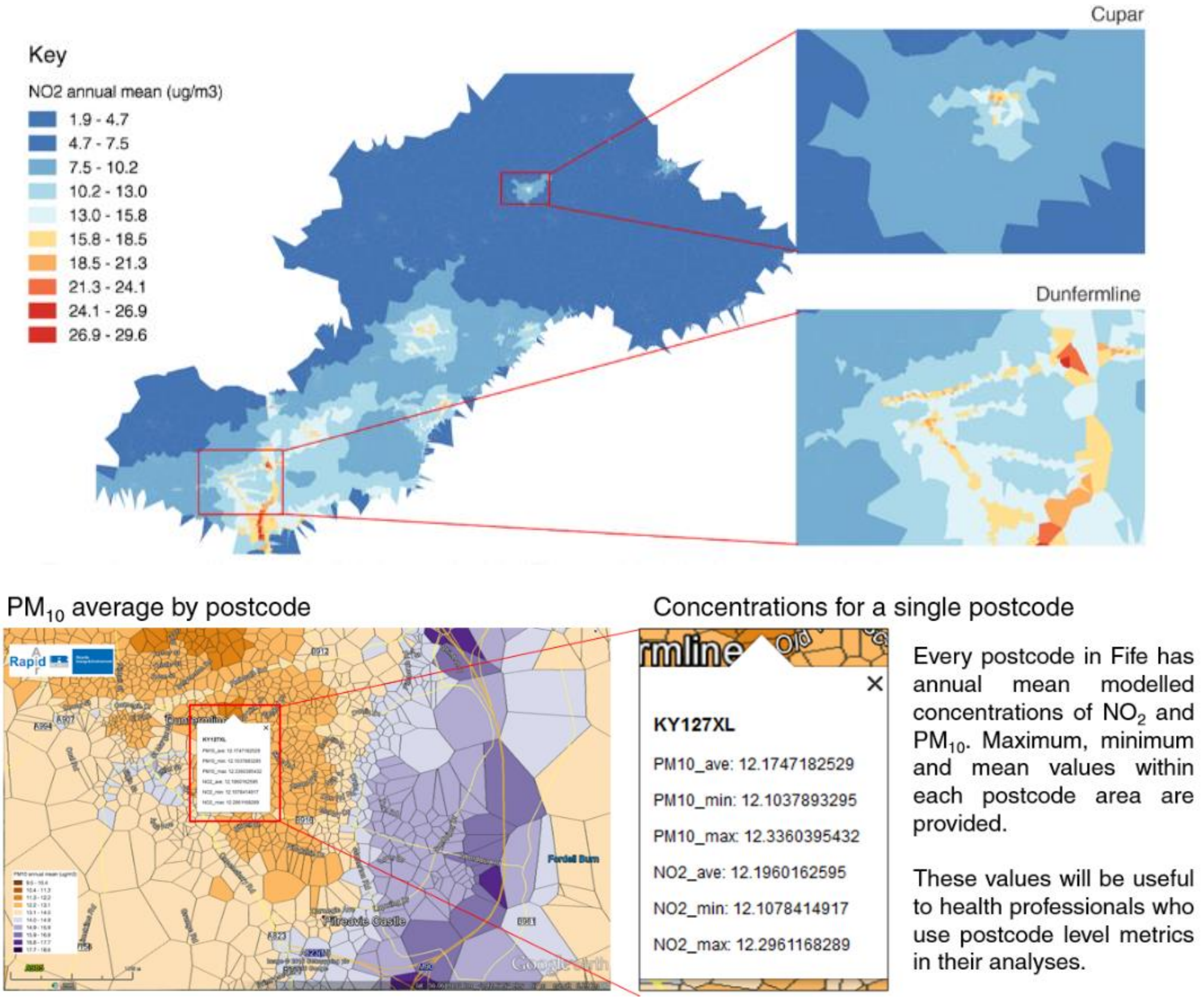
Figure 3-5 Air quality model outputs in GeoTIFF format (from RapidAIR, Greater London)



England and Wales scale | London city scale

Figure 3-6 Combining background concentrations from a regional model

Figure 3-7 Example neighbourhood analysis to support health impact studies



3.6.2 Monetising Health Impacts and Valuation

As described in detail in Section 3.5, the proposed approach for this project is to follow the IPA as detailed concentration modelling as described above is assumed to be available.

There are choices of CRFs as per listed sources in Section 3.5, however, the EC Handbook and the EEA will be followed for general EU consistency.

The concentration functions outputs will provide a change compared to a baseline scenario. Therefore, it is important to establish the baseline scenario in advance. Baseline health risk should be available in national health data statistics. This goes in line with the importance of adjusting that data to population which is to be captured. In the case of this project we would be looking at city level statistics if available.

The collection of population and its boundaries must match the concentration modelling and the boundary of air quality modelling.

The population-weighted values are multiplied by the health concentration-response functions and the values associated with each type of health impact. The difference between the baseline values and the values resulting from the CRFs can be used in the cost-benefit calculation. Splitting the results by their contributing pathways, the effects of long-term exposure on mortality rates are the dominant impact captured in the damage costs.

Likewise, the approaches to monetise health endpoints in the EC Handbook will be adopted for consistency with wider EU policy making. We propose the analysis considers the set of health impact pathways covered by the EU's Handbook on the external costs of transport[64]. Other sources (e.g. WHO, UK Defra, EEA, US BenMAP) use slight variations on the exact pathways and CRFs selected, but are broadly similar. As such in this case it is proposed to maintain consistency with the Handbook as an EU appraisal guidance document.

4 Noise

4.1 Introduction

Road traffic noise is generated primarily from a combination of the vehicle engine and friction between the vehicle and the road infrastructure. Noise is local and temporary by nature, and therefore the damage caused by noise depends on the number of people residing or otherwise being relatively close to the noise source, and the local environment which may influence dispersion.

External cost calculations usually consider two major affects when assessing noise impacts [65]:

- Health impacts: related to the long-term exposure to noise, mainly stress related health effects like hypertension and myocardial infarction.
- Lifestyle effects: relating to the disturbance which individuals experience when exposed to noise.

The top-down method is used for estimation of average costs, where detailed data is less available, and the level of generalisation is high. A typical top-down approach may use the WTP for annoyance and the environmental burden of disease methods for health effects, and multiplies these unit values with the national data on noise exposure for different noise classes. These are then divided by total kilometers travelled to produce an average impact per km driven.

The bottom-up method undertakes a more nuanced analysis considering local factors. The bottom up methodology typical follows the Impact Pathway Approach (IPA) methodology, as set out in the table below.

Table 4-1 IPA Methodology [66]

Step	Description
Noise Emissions	The changed levels of noise are measured in terms of change in time, location, frequency, level, and source of noise.
Noise Dispersion and Exposure	The differences in exposure to noise are estimated according to geographical locations and measured in dB (A) and noise level indicators. A-weighting simulates the sensitivity of the human ear, and noise levels in dB(A) resembles the relative loudness of sounds A-weighted decibels (dB(A)) are an expression of the relative loudness of sounds in air as perceived by a person.
Exposure-Response Functions	These functions present a relationship between exposure levels (in dB(A)) and negative impacts of noise. Each impact may have one or more endpoints. The response is typically described as the number or fraction of the respondents affected to a certain degree. Using the information about the number of cases of each endpoint, the overall change in noise impact is calculated.
Economic Valuation	An economic value for a unit of each endpoint of the exposure-response functions is calculated either by transferring estimates from existing valuation studies or by conducting a new original study using environmental valuation techniques.
Overall assessment	Economic value of each unit of endpoint is multiplied by the corresponding impact and aggregated over all endpoints from exposure-response functions.

A bottom-up, IPA methodology can be used to estimate marginal noise costs, which can be distinguished by three characteristics:

- Population density close to the noise source gives an indication of the population exposed to the noise. Generally, the closer to an emission source, the higher the marginal costs will be. A rough indication of the population density close to the emission source could be made by distinguishing area types (urban, suburban, rural).
- Existing noise levels (depending on traffic volume, traffic mix and speed): The higher the existing background noise level, the lower the marginal costs of an additional vehicle.
- Time of the day: according to the European Directive on environmental noise (END), noise disturbances in the evening and at night will lead to higher marginal costs than at other times of the day.

4.2 Review of methodologies

4.2.1 Noise calculation methods

A critical review of the methods to estimate the external costs of road transport noise was conducted to assess and compare the best available methodologies and their suitability for the NEMO project objectives. Seven main key papers that estimate the average and/or marginal external cost of road transport noise were identified, compared, and assessed.

There are very few recent studies providing estimates of average noise costs of transport and are generally built on earlier studies already covered in the 2004 and 2008 External Costs of Transport Handbook [67,68].

Recent estimates of marginal external noise costs are reported by Swärdh and Genell 2016 [69] and Swärdh and Genell 2020 [70].

The following table outlines the noise calculation methods used in the key reports identified.

Table 4-2 Noise calculation – best practice methodologies

Study	Coverage	Approach	Noise calculation method
Handbook on the external costs of transport [64]	EU	Top down	EEA Noise map
Update of the handbook on external costs of transport [75]	EU	Top down	EEA Noise map
External cost of transport in Europe. Update study for 2008 [68]	EU	Top down	EEA Noise map
External cost of transport – updated methodology [67]	EU	Top down	The data set of ECMT (1998) [71] is used which bases its estimates on the first UIC study [72] and on the OECD (1993) [73]
Handbook on estimation of external costs in the transport sector [74]	EU	Top down	Same as [51] above

Estimation of the marginal cost for road noise and rail noise [69]	Sweden	Bottom up	Simplified CNOSSOS-EU model for road Simplified Nord96 model for rail
Marginal costs of road noise: Estimation, differentiation, and policy implications [70]	Sweden	Bottom up	Simplified CNOSSOS-EU model for road

4.2.1.1 Noise maps

Without national empirical data or specific national model calculations, most external cost methods recommend using EU-wide noise exposure data from European Environment Agency (EEA) noise maps.

The introduction of the EEA noise maps was required by Directive 2002/49/EC (otherwise known as the Environmental Noise Directive or 'END') and provides data on exposure to noise (number of people per band of noise levels) in every agglomeration with more than 100,000 inhabitants, roads with more than 3 million vehicles per annum, railways with more than 30,000 trains per year and airports with more than 50,000 movements per year (CE Delft; INFRAS; ISI 2008). The differences in exposure to noise are estimated according to geographical locations and measured in dB(A) and noise level indicators L_{den} (day-evening-night equivalent level) and L_{night} Ricardo-AEA (2014) [75].

Information box: Methods of determining road noise to produce noise maps

Following the implementation of the END, five main methods to determine road noise have been used in Europe. The most popular method to determine noise for the EEA noise maps has been the French Nouvelle Methode de Prevision de Bruit (NMPB) 2008 method. The NMPB method calculates noise level generated by taking all of the effective factors of the physical environment, particularly the meteorological conditions, into consideration (Dutilleux, et al. (2008))76. The calculation parameters of the NMPB method are:

- Calculation per octave from 125 Hz to 4000 Hz;
- Favourable and neutral propagation paths,;
- Calculation of the percentage of favourable and neutral conditions based on the weather information;
- Characteristics of the ground in the propagation path.

The method is based on the noise propagation path principle and can be used to assess road and rail noise [77]. The method uses a large database based on readings from 41 meteorological stations across Metropolitan France, between 1987 and 2007, allowing long-term sound levels to be produced. However, the simplifications used to calculate ground attenuations in the NMPB 2008 method have now been outdated by other models based on more accurate physics (Nord2000 and Harmonoise) [78].

Three other noise mapping calculation methods used in Europe in response to END are:

- Calculation of Road Traffic Noise (CoRTN): The CoRTN method predicts noise over a 1-hour and 18-hour period at any distance up to 300m from a highway. The parameters considered in CoRTN are: traffic flow, mean speed, percentage of heavy vehicles, road surface and gradient (WSP 2014). CoRTN is a robust method and commonly used for varied terrain ranging from sparsely populated rural areas to city environments [79].
- Richtlinien für den Lärmschutz an Straßen (RLS): The RLS method requires the existence of the following input data: Hourly traffic flow average for each vehicle type; Medium speed for each

group, size, geometry and type of the road; Natural or artificial obstacles; and Main characteristics influencing the propagation of noise, such as plants, absorption from the air, reflection and diffraction [77]. An evaluation of the RLS indicated that the use of RLS allows noise levels to be predicted with good precision in areas where road noise dominates [80].

- Nordic Environmental Noise Prediction Method (Nord): The Nord method calculates noise levels corresponding to the measured noise, depending on the surrounding topography, buildings, and ground properties [69]. Predictions can be made of third-octave band levels of road traffic noise propagating over complex terrain in almost any weather condition. Thus, yearly average noise levels can be accurately computed. Nord is the most advanced engineering model although calculation speeds are slow as empirical methods or rough simplifications are not employed [81].

Due to the diversity of calculation methods used for noise mapping, comparing noise exposure levels between Member States has been challenging. The European Commission (EC) developed a fifth approach: the Common Noise Assessment Methods in Europe (CNOSSOS-EU) for noise mapping. The method provides a harmonised methodological framework for the evaluation of road, rail, aviation, and industry noise [82]. The CNOSSOS-EU method would allow the creation of a comparable database and the development of a more structured and consistent European noise policy. The EC wants the CNOSSOS-EU method to be adopted into national legislation in Member States and applied in the next round of noise mapping as required by the Environmental Noise Directive (2021/2022) (DMGR 2018). The calculation parameters, characteristic to the method, are [77]:

- Determination of noise frequency of 12 Hz to 4 kHz for road traffic noise;
- Vehicles divided into four separate categories on noise emission characterization, while the fifth category is foreseen as an open category for vehicles that can be developed in the future;
- The transmission pattern of a set of mathematical equations representing the two main sources of noise for each vehicle: rolling noise and the propulsion noise;
- The coefficients given in octave bands for each category of vehicle for a reference speed equal to 70 km/h and virtual reference road surface;
- Equivalent continuous sound pressure level at a receiver corresponding to the two types of atmospheric conditions: downward refraction and homogeneous atmospheric conditions.

The CNOSSOS-EU model is able to consider the characteristics of the environment that affect the noise level of a single vehicle, such as distance from the source, the presence of absorbing surfaces, the presence of reflective surfaces, and prevailing weather conditions. One advantage of using this model is that it can discriminate between different types of road surfaces [69]. The implementation process of the CNOSSOS-EU framework at the national level has been challenging. Deficiencies have remained, particularly in the railway model input values for sound power levels and the number of people exposed to noise [83]. Although several comparisons have concluded that the CNOSSOS-EU model does not present a considerable improvement relative to other existing models, it is important that all EU Member States adopt a common model [84].

Noise maps developed using these techniques provide data on the number of people affected by road traffic, rail traffic or aviation noise. This data can be used to estimate total and average noise costs for European countries. The noise classes that people are exposed to are classified in bins: 50-59 dB(A), 60-64 dB(A), 65-69 dB(A), 70-74 dB(A) and more than 75 dB(A). For noise levels below 55 dB(A) it is assumed no adverse effects on annoyance and health occur [80].

All noise mapping calculation methods are based on GIS modelling that combines data that includes traffic flows, noise emissions, urban mapping and population data to provide an estimate of the noise exposure in the mapped areas. However, not all data has been reported and not all cities and urban regions are included in the scope of the Environmental Noise Directive 2002/49/EC. Some validation was carried out to ensure that the predicted noise exposures were relatively accurate, however, there is a degree of uncertainty regarding the results of the noise mapping so the results must be treated with caution[80]. The most recent noise maps are for the year 2017, with data submitted up until 31/03/2017.

Open-source GIS software to produce noise maps are becoming more readily available. For example, the NoiseModelling plugin of OrbisGIS enables the production of urban noise maps based on the NMPB method for the road noise emission and using the NMPB method for the sound propagation. Without being able to simply adjust EEA noise maps, and the time and data requirements of building noise maps with open access software, another method for calculating noise exposure must be considered for this project.

4.2.1.2 Simplified noise models

To estimate the noise from road vehicles, Swärdh and Genell (2020) created a simplified variant of the CNOSSOS-EU model that is used for END noise mapping. The model assumes that the population of each locality is uniformly distributed throughout the urban areas, and that the propagation of road noise in urban areas are dependent on dwelling density and the perpendicular distance from the road.

The number of exposed individuals is then calculated by mapping the distances from the road at which a certain noise level prevails, combining the results with the length of the studied stretch of road, and finally multiplying the result by the area's population density. Traffic is calculated based on the annual average daily traffic combined with the estimated vehicle distribution over the daytime, evening, and night, the posted speed limit, and estimated differentiation between heavy, medium-heavy, and light vehicles.

The number of individuals that are exposed to a given noise level in a baseline scenario for each road section can be modelled. To calculate the marginal impact relative to this baseline, the change in the number of exposed individuals that are exposed when one vehicle is added to the current traffic is then factored in. Combining the health and disturbance costs with the change in the number of exposed individuals provides an estimate of the marginal cost of noise per vehicle kilometer for a given vehicle type and for road section [69,70].

Simplified noise models are able to reflect the latest research on health impacts and annoyance costs and have greater feasibility with existing data sets. However, the approach may not capture variables crucial in estimating noise exposure. For example, the simplified models may not consider enough categories of vehicles and road surfaces, and do not include a suitable approach for the stop and start behaviour of drivers and the influence of slopes. Current models also have a poor capacity to estimate noise from vehicles moving under 30 km/h, which is a common speed in urban areas [77]. Without access to a simplified noise model, an alternative method for estimating noise exposure must be considered.

4.2.2 Noise impacts and valuation methods

The exposure to noise results in several impact endpoints due to prolonged and frequent exposure to transport noise. The following impacts have significant evidence [68,82,86,93]:

- ischaemic heart diseases;
- stroke;
- dementia;
- hypertension;

- annoyance.

The UK Interdepartmental Group on Costs and Benefits Noise subject group (IGCBN) established the use of dose-response functions as a means to quantify the ***health impacts*** of noise [85].The function relates the exposure to a certain noise level to an end effect (such as stroke) via a relative risk.

The WHO [86] apply a similar methodology based on the environmental burden of disease, which combines exposure-response relationships, exposure distribution, background prevalence of disease and disability weights of the outcome. The methodology calculates the burden of disease in terms of disability-adjusted life-years (DALYs).

Alternative methods to value health impacts use the costs of a premature death and medical costs of disease that include the costs of hospital and absentee costs [68,89,86, 93].

For road traffic it may be difficult to separate the health effects of emissions and noise since they are strongly correlated [87].

Annoyance costs are based on individual preferences using stated or revealed preference methods. The hedonic pricing approach is the most common revealed preference (RP) method and utilises the reduction in property values resulting from noise exposure as a proxy for noise costs. Stated preferences (SP) methods use questionnaires or experiments where respondents are asked to provide their WTP (or WTA) to avoid noise from transport. SP methods can take two forms: contingent valuation through the use of questionnaires or surveys, and choice experiments.

The RP method relies on actual market behaviour, where stated preference techniques (e.g. individuals' WTP for avoiding transport noise) can be surveyed. However, the results from the RP approach are sensitive to the conditions of the markets observed. SP methods directly measure the WTP and they also allow the researcher to control for all external factors, to isolate the impact of noise. Nonetheless, SP methods depend very much on the survey / experiment design and the level of information, struggles to avoid strategic behaviour of respondents and only involves hypothetical expenditures [64]. In both cases, a lack of knowledge of the market actors on the damage caused by transport noise may seriously affect the reliability of the stated preference results [64].

Table 4-3 outlines the noise valuation methods used in the key reports identified, and some of the key impact parameters applied.

The average noise cost methodology adopted in the latest EU handbook on the external costs of transport [64] multiplies together two input values: number of people exposed to noise for each transport mode, and the noise costs per person exposed. Summing these costs together gives the total external noise costs for a transport mode. The total costs are allocated to specific vehicle categories based on weighting factors. Finally, average noise costs are estimated by dividing the total costs by vehicle-kilometre (vkm) or tonne-kilometre (tkm).

Table 4-3 Noise valuation methods – best practice methodologies

Study	Coverage	Approach
Handbook on the external costs of transport [64]	EU	Annoyance value is calculated using a WTP approach. The health value is based on an environmental burden of disease method and are taken from DEFRA (2014) [88].
Handbook on estimation of external costs in the transport sector [74]	EU	Annual WTP-value equal to 0.09% – 0.11% of capita income per dB.

		Health costs are based on medical costs and a value of €50,000 – 75,000 for a life year lost which corresponds to the research by ExternE [89].
External cost of transport in Europe. Update study for 2008 [68]	EU	The costs of annoyance are based on stated-preference research by (Navrud 2002) [66]. To estimate the health costs a distinction was made between medical costs and costs of premature deaths. To value the latter, a value of a life year lost of € 40,300 (price base 2000) is used. The medical costs include the costs of the hospital and absentee costs.
Update of the handbook on external costs of transport [75]	EU	Adopted the approach used by the 2008 Handbook and updated in 2011.
External cost of transport – updated methodology [67]	EU	WTP for silent space above 55 dB(A) and medical costs of diseases, caused by transport noise exposure.
Estimation of the marginal cost for road noise and rail noise [69]	Sweden	Disturbance costs are short-term costs based on recent Swedish hedonic valuations of traffic noise with separate studies for rail noise [90], and road noise [91], respectively. Health impact costs are mainly taken from the Swedish Transport Administration (Swedish Transport Administration 2016) [92], (Ricardo-AEA 2014)[75], and (Holland 2014) [93].
Marginal costs of road noise: Estimation, differentiation, and policy implications [70]	Sweden	Same as [69] above.

4.3 Discussion and recommended methodology

A detailed review of approaches to estimating the external costs of noise exposure has been undertaken and highlights that this assessment is typical undertaken either: (a) following a top-down approach using noise maps to calculate average costs or (b) following a bottom-up approach using detailed noise models to calculate marginal costs.

For this project, external costs will be calculated associated with the novel techniques being explored, as such we are most interested in assessing the marginal impacts of such techniques to compare to the costs. However, detailed noise modelling is not possible within the bounds of this project. Furthermore, there limitations around the extent to which noise maps can be easily manipulated to test the marginal impact of specific measures.

Hence for this project, an appraisal methodology is adopted based on existing appraisal techniques that can be readily applied or easily adopted without access to more detailed models. This approach varies depending on the type of measure under considerations, as different measures may have different effects. The distinction is as follows:

1. Measures affecting vehicle flows: e.g. restricting high noise-emitting vehicles from entering a given zone or using a given road.
2. Measures affecting noise generation or dispersion (but without affecting vehicle flows): e.g. noise barriers, or road surface measures.

4.3.1 Measures affecting vehicle flows

Policy options may reduce the exposure to road transport noise by decreasing traffic. The value of traffic noise reduction in this case can be estimated using a simpler top-down methodology. Marginal noise costs from the EC Handbook on the external costs of transport [64] can be multiplied by the reduction in total distance travelled for each vehicle category as a result of the policy option to value the change in noise. Real-world noise measurements using a novel autonomous remote sensing device (N-RSD) developed under the NEMO project will be used to sense check and validate the noise costs applied and the assumptions underpinning them. The array of microphones will provide important measurements of noise levels from **individual vehicles**, which can then be aggregated into different vehicle categories. Options to use the N-RSD measurements further in the calculations will be explored as the study develops and results begin to emerge. Table 4-4 from the EC Handbook on the external costs of transport provides an example of the marginal costs of road transport that can be applied.

Table 4-4 Marginal noise costs road transport – in EUR-cent (2016) per pkm (data for 2016) [64]

Road	Time of the day	Traffic situation	Urban	Suburban	Rural
Passenger transport (EUR cent per km)					
Passenger car	Day	Dense	0.5	0.03	0.004
		Thin	1.1	0.07	0.009
	Night	Dense	0.9	0.05	0.007
		Thin	2.1	0.13	0.015
Bus	Day	Dense	0.05	0.03	0.004
		Thin	1.3	0.08	0.010
	Night	Dense	1.0	0.05	0.008
		Thin	2.4	0.15	0.018

4.3.2 Measures affecting noise generation or dispersion

The study will also explore a set of technologies which will not impact on vehicle flows, but will still affect noise generation and dispersion (e.g. noise barriers). In this case, given the technology does not affect vehicle flows so the marginal costs from the Handbook (which are expressed per change in vkm) cannot be applied. Hence in this case a different approach has been developed, based on the methodologies previously applied in the literature. This adopts a step-wise approach, akin to the IPA, but with important simplifying assumptions at different steps to overcome the lack of access to detailed noise models. The method provides flexibility in studying different mix of vehicles on the road network and allows for potential changes in noise arising from policy or technology to be investigated.

4.3.2.1 Noise impact and exposure

The external cost of transport noise will depend on the number of people effected and the type of road in close proximity to where the inhabitants live. The L_{den} metric is the EU policy indicator, whereby people exposed to 55 dB or more are considered to be affected by noise.

Impacts of various road fabric measures on noise will be adopted from a previous study by Milford et al. (2012) [94] (see Table 4-5).

Table 4-5 Estimated noise reduction by measure [94]

Measure	Noise reduction (dB)
Noise barrier	1 - 8
Façade insulation	8
Porous double layer	4
Porous single layer	2
Thin layer asphalt	2

One of the objectives of the NEMO project is to explore and test novel asphalt materials and multifunction barriers, with the aim of mitigating noise and emissions from on-road vehicles. The laboratory and real-life performance noise measurements of these abatement technologies will provide important information which can be used to validate and update the estimated noise reduction values provided in Table 4-5.

To quantify the dispersion and exposure of these changes in noise levels, assumptions will be adopted (again from the literature) regarding the numbers of people potentially affected. Table 4-6 illustrates the variation between different road categories and the number of inhabitants per kilometer along each road category. The number of exposed people per kilometer are estimated by Roo et al. (2011) [95] from noise maps and demographic data. The data in Table 4-6 is the basis for calculating how many people will be affected by a road transport noise reducing measure.

Table 4-6 Overview of road sub types and exposed inhabitants [95]

Road type	Residential urban/suburban	Residential urban/suburban	Main roads urban/suburban	Main roads urban/suburban	Arterial roads urban/suburban	Urban motorways urban/suburban	Rural motorways	Rural roads
Traffic type	Intermittent	Free flow	Intermittent	Free flow	Free flow	Free flow	Free flow	Free flow
Speed range (km/h)	<50	<50	<50	<50	50-70	70-120	80-130	50-100
Estimate average exposed inhabitants (per km)	250	250	500	500	500	1000	40	20

Data from the latest road noise maps are used in this methodology will then be used to distribute the number of people exposed to each noise band. Without access to detailed noise maps and models, simple adjustments can be made to the number of people exposed to the median of each noise band based on the estimated noise reductions of each measure [94]. The median of the >75 noise band will assumed to be 77 dB. The change in the number of exposed individuals in each noise band when a noise reduction policy is introduced will be used in the final calculation.

4.3.2.2 Monetising Health Impacts and Valuation

The annoyance costs will be based (where possible) on a recent local hedonic valuation study of road noise where housing prices for differently noise-exposed properties are analysed. The hedonic function should be based on the full demand for noise abatement estimated by estimating the marginal WTP for noise abatement beyond endogeneity issues and by controlling for other factors influencing the housing price. If hedonic functions are not available, the annoyance costs per dB based on the most recent insights by Bristow, Wardman and Chintakayala (2015) [96] should be used.

The source of health impact costs for noise are from WHO (2011) [86], and ExternE project (Hunt, 2005) [89]. The outcomes of these health impacts are in: lost life years; number of non-fatal cases; days of hospitalisation; days of work absence; and days of illness. The impact functions are dependent on the noise level. For each additional unit of noise exposure, the marginal effect from WHO (2011) [86], is used and related to the number of individuals exposed to the noise level along the various road sections in the data.

The marginal effect of a myocardial infarction specifically considers the base risk (BR) and relative risk (RR). The RR is expressed by risk ratios attributed to different L_{den} categories in Table 4-7 below. To calculate the BR for myocardial infarction (years of lost life), the number of fatal myocardial infections in a year is divided by the total national population in the same year.

Table 4-7 Exposure–response curve (polynomial fit) of the association between traffic noise and incidence of myocardial infarction (WHO 2011) [86]

L_{day}	L_{den}	RR		L_{day}	L_{den}	RR
55	57	1		**68**	70	1.108
56	58	1		**69**	71	1.128
57	59	1		**70**	72	1.149
58	60	1.003		**71**	73	1.173
59	61	1.007		**72**	74	1.198
60	62	1.012		**73**	75	1.225
61	63	1.015		**74**	76	1.254
62	64	1.027		**75**	77	1.285
63	65	1.036		**76**	78	1.391
64	66	1.047		**77**	79	1.354
65	67	1.06		**78**	80	1.391

66	68	1.074		**79**	81	1.431
67	69	1.091		**80**	82	1.473

Regarding the impact functions of ExternE, these are not dependent on the noise level, except for the threshold value of 70 dB L_{den}. Further, regarding the impact functions taken from ExternE are given per 1,000 of exposed adults. Thus, these functions should be divided with the factor calculated as the share of all individuals that were 18 years and older (AF). The impact functions to be deployed are set out in the following table.

Table 4-8 Impact functions for the health costs of noise exposure

Impact and unit	Source	Marginal effect per 1000 adults exposed to noise >70 dB, L_{den}	Marginal effect per 1000 individuals exposed to noise >55 dB, L_{day}
Myocardial infarction, years of lost life	WHO, 2011		13.2 x BR x RR
Myocardial infarction, days of hospitalisation	WHO, 2011		18 x BR x RR
Myocardial infarction, days of work absence	WHO, 2011		320 x BR x RR
Myocardial infarction, no. of non-fatal cases	WHO, 2011		2.079 x RR
Angina pectoris, days of hospitalisation	Hunt, 2005	0.168 / AF	
Angina pectoris, days of work absence	Hunt, 2005	0.684 / AF	
Days of work absence, days of illness	Hunt, 2005	0.240 / AF	
High blood pressure, days of hospitalisation	Hunt, 2005	0.063 / AF	

To produce a marginal cost of noise, we must attach a monetary value to each health outcome. These valuations are presented in Table 4-9 below, updated to the 2021 price year based on the eurozone inflation rate. These existing valuation studies provide an economic value for a unit of each endpoint of the exposure-response functions. The economic value of each unit of endpoint is multiplied by the corresponding impact and aggregated over all endpoints from the impact functions.

Table 4-9 Valuation of transport noise health effects; in EUR, price year 2021

Transport noise effect	Source	Unit	Valuation in EUR
Premature deaths	Handbook on the external costs of transport [64]	Per lost life year (VOLY)	73,500
Symptom myocardial infarction	Handbook on estimation of external costs in the transport sector [74]	Per non-fatal case	31,610
Symptom angina pectoris	Handbook on estimation of external costs in the transport sector [74]	Per day of illness	19,809
Work absence	Holland, 2014 [93]	Per day	163
Health care costs	Holland, 2014 [93]	Per hospitalised day	2,788

4.3.2.3 Calculation of estimated marginal costs

The equation below has been adapted from the estimation of the marginal cost for road noise in Sweden by Swärdh and Genell (2020)[70].

$$C_{ik} = \sum_{55-59}^{>75} \Delta N_i^{Lden} \times HF_k \times D_i^{-1} \times 365^{-1} + \sum_{j=1}^{J} \sum_{L=55-59}^{>75} ME_{jL} \times \Delta N_i^{Lden} \times V_{jL} \times D_i^{-1} \times 365^{-1}$$

The external cost of road transport noise consists of two additive terms: the top section captures the annoyance costs, and the bottom section captures the health costs.

L_{den} is the noise level in dB; ΔN is the change in the number of exposed individuals when a noise reduction policy is introduced; HF is the hedonic price function; D is the length in kilometres of the considered road section; j represents the health impacts outlined in Table 4-8; ME_L is the marginal effect of the health impacts; and V is the monetary valuation of the health outcomes presented in

Table 4-9. Finally, we need to adjust the marginal calculations by the number of days, otherwise the result will be the marginal cost of a vehicle kilometre driven each of the 365 days per year. Note that ME_L is zero for noise levels below 55 dB. Note also that 55-59 dB is the lowest noise band and >75 the highest noise band.

Conclusions

This report provides a review of methodologies and guidance to appraise the impact of pollutant emissions and noise levels from road and rail, on human health and the environment. Taking into consideration the aims of the NEMO project we have provided recommended methodologies for emission and noise (Section 3.6 and 4.3) based on the Impact-Pathway Approach. These approaches will result in a set of damage costs associated with a marginal change in pollutant or noise emissions (expressed in €/ton of pollutant emitted or change in cost of noise per vehicle kilometre for a given vehicle type and for road/rail section).

The damage cost values are an estimate of the societal cost that is associated with a marginal change in pollutant or noise emissions. Combining the estimated costs with forecasts of changes associated with mitigation techniques, costs, and benefits of mitigation scenarios, relative to the baseline, can be evaluated.

The report also explores the use of remote sensing technologies to acquire empirical data from road traffic pollutant and noise emissions for different vehicle categories, in order to improve the external cost estimations. The remote sensing measurements of pollutants from vehicle exhausts can be used to improve or calculate emission factors to be used in the air quality modelling strategies. For the external cost estimations associated with noise, the data from the noise emissions remote sensing device may be used to sense check and validate the EC handbook costs applied in the proposed methodology, and the assumptions underpinning them.

This approach shows the value of NEMO as a whole and demonstrates how the project strategy is directly linked to policies to control and improve air quality in urban areas.

Bibliography

[1] Cimorelli, A., Perry, S., Venkatram, A., Weil, J., Paine, R., Wilson, R., Lee, R., Peters, W. and Brode, R., (2005). AERMOD: A Dispersion Model for Industrial Source Applications. Part I: General Model Formulation and Boundary Layer Characterization. Journal of Applied Meteorology, 44(5), pp.682-693.

[2] Brode, R.W., (2005). AERMOD: A dispersion model for industrial source applications. Part I: general model formulation and boundary layer characterization. Applied Meteorology 44, 682-693.

[3] Masey, N., Hamilton, S., & Beverland, I. (2018). Development and evaluation of the RapidAir® dispersion model, including the use of geospatial surrogates to represent street canyon effects. Environmental Modelling and Software, 108, 11.

[4] Buckland, A. and Middleton, D. (1999). Nomograms for calculating pollution within street canyons. Atmospheric Environment, 33(7), pp.1017-1036.

[5] Snyder, M.G. and Heist, D.K., 2013. User's Guide for R-LINE Model Version 1.2. A Research LINE source model for near-surface releases. USEPA/ORD/NERL MD-81. Research Triangle Park. United States of America.

[6] Qian, W. and Venkatram, A., (2011). Performance of steady-state dispersion models under low wind-speed conditions. Boundary-Layer Meteorology 138, 475-491.

[7] Defra. (2018). Local Air Quality Management - Technical Guidance TG16. Defra.

[8] European Commission (2020) INSPIRE KNOWLEDGE BASE (https://inspire.ec.europa.eu/id/document/tg/tn)

[9] Open Transport Map (2015) http://opentransportmap.info/

[10] OpenStreetMap (2020) https://www.openstreetmap.org/about

[11] Eurogeographics (2021) https://eurogeographics.org/open-maps-for-europe/

[12] TomTom (2020) TomTom Traffic Index https://www.tomtom.com/en_gb/traffic-index/

[13] Ricardo Energy & Environment (2018) Southampton Clean Air Zone - Air Quality Modelling Report (AQ2)

[14] European Data Portal (2021) https://www.europeandataportal.eu/data/datasets/lidar-point-cloud?locale=en

[15] COPERT https://www.emisia.com/utilities/copert/

[16] The Handbook Emission Factors for Road Transport (HBEFA) https://www.hbefa.net/e/index.html

[17] EEA (2019) Europe's urban air quality — re-assessing implementation challenges in cities

[18] Matthias, Volker, Johannes Bieser, Tudor Mocanu, Thomas Pregger, Markus Quante, Martin OP Ramacher, Stefan Seum, and Christian Winkler (2020): Modelling road transport emissions in Germany–Current day situation and scenarios for 2040. Transportation Research Part D: Transport and Environment 87 102536.

[19] Rodriguez-Rey, Daniel, Marc Guevara, Ma Paz Linares, Josep Casanovas, Juan Salmerón, Albert Soret, Oriol Jorba, Carles Tena, and Carlos Pérez García-Pando (2021) :A coupled macroscopic traffic and pollutant emission modelling system for Barcelona. Transportation Research Part D: Transport and Environment 92, 102725.

[20] Santos, G. Sousa, Ingrid Sundvor, Matthias Vogt, Henrik Grythe, Tormod Wergeland Haug, B. A. Høiskar, and Leonor Tarrasón (2020): Evaluation of traffic control measures in Oslo region and its effect on current air quality policies in Norway. Transport Policy 99, 251-261.

[21] Senate Department for Urban Development and the Environment (2014): Air quality plan for Berlin 2011-2017. https://www.berlin.de/senuvk/umwelt/luftqualitaet/de/luftreinhalteplan/download/lrp_150310_en.pdf

[22] Ferreira, F., P. Gomes, H. Tente, A. C. Carvalho, P. Pereira, and J. Monjardino (2015): Air quality improvements following implementation of Lisbon's Low Emission Zone. Atmospheric Environment 122, 373-381.

[23] European Data Portal https://www.europeandataportal.eu/data/datasets/vm-uvek-shapefile-2020-bundesamt-fur-raumentwicklung-are?locale=en

[24] EU Open Data Portal https://data.europa.eu

[25] GraphHopper Open Traffic Collection (GitHub page) https://github.com/graphhopper/open-traffic-collection

[26] Matthias, Volker, Johannes Bieser, Tudor Mocanu, Thomas Pregger, Markus Quante, Martin OP Ramacher, Stefan Seum, and Christian Winkler (2020): Modelling road transport emissions in

Germany–Current day situation and scenarios for 2040. Transportation Research Part D: Transport and Environment 87, 102536.

[27] Rodriguez-Rey, Daniel, Marc Guevara, Ma Paz Linares, Josep Casanovas, Juan Salmerón, Albert Soret, Oriol Jorba, Carles Tena, and Carlos Pérez García-Pando. A coupled macroscopic traffic and pollutant emission modelling system for Barcelona (2021): Transportation Research Part D: Transport and Environment 92, 102725.

[28] Santos, G. Sousa, Ingrid Sundvor, Matthias Vogt, Henrik Grythe, Tormod Wergeland Haug, B. A. Høiskar, and Leonor Tarrasón (2020): Evaluation of traffic control measures in Oslo region and its effect on current air quality policies in Norway. Transport Policy 99, 251-261.

[29] Gastaldi, Massimiliano, Riccardo Rossi, Gregorio Gecchele, and Luca Della Lucia (2013): Annual average daily traffic estimation from seasonal traffic counts. Procedia-Social and Behavioral Sciences 87, 279-291.

[30] TomTom (2020) https://www.tomtom.com/products/road-traffic-data-analytics/

[31] Teletrac Navman (2020) https://www.teletracnavman.co.uk/

[32] https://www.basemap.co.uk/trafficmaster-data/

[33] Christodoulou A., Christidis P (2020): Measuring congestion in European cities: A focus on Brussels, Seville and Krakow, JRC Working Papers JRC118448, Joint Research Centre (Seville site)

[34] HBEFA: https://www.hbefa.net/e/index.html

[35] Preiss, P. and Klotz, V. (2007): Description of updated and extended draft tools for the detailed site-dependent assessment of external costs. Technical Paper No. 7.4 - RS 1b of NEEDS Project.

[36] WHO (2013): Health risks of air pollution in Europe – HRAPIE project Recommendations for concentration–response functions for cost–benefit analysis of particulate matter, ozone and nitrogen dioxide.
https://www.euro.who.int/__data/assets/pdf_file/0006/238956/Health_risks_air_pollution_HRAPIE_project.pdf

[37] WHO (2013a): European mortality database (MDB) [online]. Copenhagen, WHO Regional Office for Europe.

[38] Hoek G et al. (2013) : Long-term air pollution exposure and cardio-respiratory mortality: a review. Environmental Health, 12:43.

[39] IHME (2013) : Global Burden of Disease Study 2010 (GBD 2010) – ambient air pollution risk model 1990–2010. Seattle, WA, Institute for Health Metrics and Evaluation

[40] WHO (2013b) : European detailed mortality database [online]. Copenhagen, WHO Regional Office for Europe

[41] WHO (2013c) :European health for all database [online]. Copenhagen, WHO

[42] Woodruff TJ, Grillo J, Schoendorf KC (1997) : The relationship between selected causes of postneonatal infant mortality and particulate air pollution in the United States. Environmental Health Perspectives, 105(6):608–12.

[43] Hoek G et al. (2012): PM_{10}, and children's respiratory symptoms and lung function in the PATY study. European Respiratory Journal, 40(3):538–547.

[44] WHO (2013d): European mortality database (MDB) [online]. Copenhagen, WHO Regional Office for Europe

[45] WHO (2013e): European hospital morbidity database [online]. Copenhagen, WHO Regional Office for Europe

[46] Ostro BD, Rothschild S (1989): Air pollution and acute respiratory morbidity: an observational study of multiple pollutants. Environmental Research, 50:238–247.

[47] Ostro BD (1987): Air pollution and morbidity revisited: a specification test. Journal of Environmental Economics Management, 14(1):87–98.

[48] Lai CK et al. (2009): Global variation in the prevalence and severity of asthma symptoms: phase three of the International Study of Asthma and Allergies in Childhood (ISAAC). Thorax, 64(6):476–483.

[49] Weinmayr G et al. (2010): Short-term effects of PM10 and NO2 on respiratory health among children with asthma or asthma-like symptoms: a systematic review and meta-analysis. Environmental Health Perspectives, 118:449–457.

[50] Migliore E et al. (2009): Respiratory symptoms in children living near busy roads and their relationship to vehicular traffic: results of an Italian multicenter study (SIDRIA 2). Environmental Health, 8:27.

[51] McConnell R et al. (2003): Prospective study of air pollution and bronchitic symptoms in children with asthma. American Journal of Respiratory and Critical Care Medicine, 168(7):790–797

[52] WHO (2013f): Review of evidence on health aspects of air pollution – REVIHAAP project: technical report. Copenhagen, WHO Regional Office for Europe

[53] Air Quality damage cost update 2019: report for DERFA. https://uk-air.defra.gov.uk/assets/documents/reports/cat09/1902271109_Damage_cost_update_2018_FINAL_Issue_2_publication.pdf

[54] Holland, M., A. Wagner, T. Davies, J. Spadaro, and M. Adams. (2011): Revealing the costs of air pollution from industrial facilities in Europe. https://www.eea.europa.eu/publications/cost-of-air-pollution

[55] Sullivan, Patrick W., Julia F. Slejko, Mark J. Sculpher, and Vahram Ghushchyan (2011): Catalogue of EQ-5D scores for the United Kingdom. Medical Decision Making 31, no. 6, 800-804.

[56] WHO (2018): Health statistics and information systems, Software tools, DISMOD II. https://www.who.int/healthinfo/global_burden_disease/tools_software/en/

[57] EMEP/EEA air pollutant emission inventory guidebook 2019 https://www.eea.europa.eu/themes/air/air-pollution-sources-1/emep-eea-air-pollutant-emission-inventory-guidebook

[58] https://www.eea.europa.eu/publications/emep-eea-guidebook-2019/part-b-sectoral-guidance-chapters/1-energy/1-a-combustion/1-a-3-b-i/view

[59] https://www.eea.europa.eu/publications/emep-eea-guidebook-2019/part-b-sectoral-guidance-chapters/1-energy/1-a-combustion/1-a-3-b-i/at_download/file

[60] https://www.eea.europa.eu/publications/emep-eea-guidebook-2019/part-b-sectoral-guidance-chapters/1-energy/1-a-combustion/road-transport-appendix-4-emission/at_download/file

[61] World Meteorological Organization https://public.wmo.int/en

[62] Meteorological Monitoring Guidance for Regulatory Modeling Applications (EPA-454/R-99-005) https://www.epa.gov/sites/production/files/2020-10/documents/mmgrma_0.pdf

[63] User's Guide for the AERMOD Meteorological Preprocessor (AERMET) https://gaftp.epa.gov/Air/aqmg/SCRAM/models/met/aermet/aermet_userguide.pdf

[64] EC Handbook on the external costs of transport. https://op.europa.eu/en/publication-detail/-/publication/9781f65f-8448-11ea-bf12-01aa75ed71a1

[65] Updating Marginal External Costs of Road Freight Aktins, Jacobs. https://assets.publishing.service.gov.uk/government/uploads/system/uploads/attachment_data/file/865553/marginal-external-costs-technical-report.pdf

[66] Navrud, S., (2002): The state-of-the-art on economic valuation of noise. Final Report to European Commission DG Environment, 14.

[67] INFRAS, IWW (2004): External costs of transport in Europe.

[68] DE Delft INFRAS Fraunhofer ISI (2011):, External Costs of Transport in Europe Update Study for 2008

[69] Swärdh, Jan-Erik, and Anders Genell. (2016): Estimation of the marginal cost for road noise and rail noise. VTI notat 22A-2016

[70] Swärdh, Jan-Erik, and Anders Genell (2020): Marginal costs of road noise: Estimation, differentiation and policy implications. Transport Policy 88, 24-32.

[71] ECMT (1998): Efficient Transport for Europe, Policies for Internalisation of External costs, Paris 1998.

[72] INFRAS, IWW (1995): External effects of transport, UIC, Karlsruhe, Zurich, Paris 1995

[73] OECD (1993): Environmental Data - Donnees OCDE sur L'environnement, Compendium 1993, Paris 1993

[74] Maibach, Markus, Christoph Schreyer, Daniel Sutter, H. P. Van Essen, B. H. Boon, Richard Smokers, Arno Schroten, C. Doll, Barbara Pawlowska, and Monika Bak (2008): Handbook on estimation of external costs in the transport sector. *Ce Delft* 336

[75] Ricardo AEA (2014): Update of the Handbook on External Costs of Transport, Report for the European Commission: DG MOVE. https://ec.europa.eu/transport/sites/transport/files/handbook_on_external_costs_of_transport_2014_0.pdf

[76] Dutilleux, Guillaume, Jero me Defrance, Benoit Gauvreau, and Francis Besnard (2008): The revision of the French method for road traffic noise prediction. *Journal of the Acoustical Society of America* 123, no. 5, 3150-3150.

[77] Petrovici, Alina, Claudia Tomozei, Florin Nedeff, Oana Irimia, and Mirela Panainte-Lehadus. Review on the road traffic noise assessment. Journal of Engineering Studies and Research 22, no. 1 (2016): 81-89.

[78] Xuetao Zhang (2014): Three Typical Noise Assessment Methods in EU, SP Report 2014:33, June 15, 2014.

[79] Kean, S., (2008): Is CoRTN an Leq or L10 procedure. Proc Australian Acoustics Society ‘08, pp.24-26.

[80] Murillo-Gómez, Diego Mauricio, Juan Camilo Gil-Carvajal, Valentina Zapata-Rodríguez, and José Julián Téllez-García (2015): Assessment of the RLS 90 calculation method for predicting road traffic noise in Colombian conditions. Revista Facultad de Ingeniería Universidad de Antioquia 75, 175-188.

[81] Kragh, Jørgen (2011): Traffic noise prediction with Nord2000-an update. In Proceedings of ACOUSTICS.

[82] Kephalopoulos, Stylianos, Marco Paviotti, and Fabienne Anfosso-Lédée (2012): Common noise assessment methods in Europe (CNOSSOS-EU). Common noise assessment methods in Europe (CNOSSOS-EU), 180-p.

[83] Kokkonen, Jarno (2018): CNOSSOS-EU noise model implementation in Finland and experience of it in 3rd END round. In Proceedings of the Euronoise, pp. 27-31

[84] Ribeiro, João Nuno Esteves Calado (2016): Assessment of the CNOSSOS-EU model for road traffic noise prediction, Tecnico Lisboa, Lisboa.

[85] Interdepartmental Group on Costs and Benefits Noise Subject Group (IGCB(N)) Second report (2010): Valuing the human health impacts of environmental noise exposure

[86] WHO (2011). Burden of disease from environmental noise: Quantification of healthy life years lost in Europe. World Health Organization. Regional Office for Europe.

[87] Beelen, Rob, Gerard Hoek, Danny Houthuijs, Piet A. van den Brandt, R. Alexandra Goldbohm, Paul Fischer, Leo J. Schouten, Ben Armstrong, and Bert Brunekreef (2009): The joint association of air pollution and noise from road traffic with cardiovascular mortality in a cohort study. Occupational and environmental medicine 66, no. 4,243-250.

[88] DEFRA (2014): Environmental Noise: Valuing impacts on: sleep disturbance, annoyance, hypertension, productivity and quiet. https://assets.publishing.service.gov.uk/government/uploads/system/uploads/attachment_data/file/380852/environmental-noise-valuing-imapcts-PB14227.pdf

[89] Hunt, A. (2005): Impact Pathway Approach: Monetary Valuation. In ExternE Externalities of Energy: Methodology 2005 Update, pp. 133-180. Office for Official Publications of the European Communities.

[90] Jan-Erik Swärdh, Henrik Andersson, Lina Jonsson, and Mikael Ögren (2012): Estimating non-marginal willingness to pay for railway noise abatement: Application of the two-step hedonic regression technique. Center for Transport Studies Working Paper 2012:27

[91] Andersson, Henrik, Jan-Erik Swärdh, and Mikael Ögren (2013): Efterfrågan på tystnad: skattning av betalningsviljan för icke-marginella förändringar av vägtrafikbuller.

[92] Bångman, G., and P. Nordlöf (2016): Analysmetod och samhällsekonomiska kalkylvärden för transportsektorn: ASEK 6.0.

[93] Holland, Mike (2014): Cost–benefit Analysis of Final Policy Scenarios for the EU Clean Air Package.

[94] Milford, Ingunn, Sigve J. Aasebo, and Kjell Strommer (2012): Value for money in road traffic noise abatement. Procedia-Social and Behavioral Sciences 48, 1366-1374.

[95] Roo, F. de, M. G. Dittrich, P. J. G. van Beek, C. Bosschaart, Derksen DG, and M. de Kievit (2011): VENOLIVA-Vehicle Noise Limit Values-comparison of two noise emission test methods. Final Report.

[96] Bristow, Abigail L., Mark Wardman, and V. Phani Kumar Chintakayala (2015): International meta-analysis of stated preference studies of transportation noise nuisance. Transportation 42, no. 1 71-100.