Report

Comparison of the impact of rail and road freight on local air quality

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Executive summary

This report has been commissioned to compare the effects of transporting freight by road and rail on local air quality. The report has been prepared to be accessible to a wide audience, including rail industry sustainability staff, policy makers, decision makers, and technical practitioners. Technical terminology is used where necessary and accompanied by concise explanations to aid interpretation.

Air quality is a result of multiple factors that cause it to vary from location to location. The total mass of emissions that are released is just one of these factors. The dispersion and atmospheric processes affecting emissions once they have been released determines the concentration of pollutants that a person breathes in at a particular location, so this is just as important as the amount of emissions that are released.

Rail and road emissions were modelled for three representative routes, covering a mixture of traffic type, origins and destinations, and proportion of urban areas traversed. The equivalent number of heavy goods vehicle (HGV) journeys necessary to carry the same train load were considered, along with 'Last Mile' emissions for an HGV journey from the rail terminal to the final destination where relevant. Return trips were considered in all cases. The representative routes were:

- Tunstead – Battersea (aggregates): one rail service equivalent to 91 HGVs
- Merehead – Acton – Bow (aggregates): one rail service equivalent to 76 HGVs
- London Gateway – Middlemore Industrial Estate, West Midlands (intermodal): three services per day, each equivalent to 50 HGVs.

This emissions modelling provided a realistic comparison (on a gram-per-tonne-kilometre basis) of road and rail emissions along the full route. For carbon dioxide (CO_2), rail emissions are significantly lower than road emissions, particularly for the aggregate flows, reflecting the energy efficiency of steel wheel on steel rail. For emissions of particulate matter (PM), rail performs better than road, at least for newer locomotive types. However, for emissions of nitrogen oxides (NOx), rail performs worse than road, reflecting the age of locomotives and the faster turnover of the HGV fleet, which has introduced HGVs that meet the Euro VI emissions standard.

For this pilot study, dispersion modelling was then used to estimate how emissions react and disperse in the atmosphere and thereby affect the concentration of pollutants experienced at residential locations. ADMS-Roads software was used. This is a state-of-the-art model that is widely used for assessing and managing local air quality. A number of modelling simulations were carried out, representing a range of typical conditions, building configurations, and different vehicle types and speeds. This enabled the development of dispersion fall-off curves for the concentrations of air pollutants to be defined. These curves identified the level of emissions that would be experienced at varying distances away from the road or railway line, depending on the speed of the vehicle and local conditions. This approach enabled a comparative assessment of the potential change in air quality from road or rail freight.

Finally, for the three representative routes, the dispersion modelling outputs were combined with GIS-based population density data and building configurations to estimate the increase in population exposure to pollution from road versus rail freight.

The results of this study provide the rail industry with key learnings about rail freight and urban air quality:

- The impact of a typical daily rail freight service or the equivalent number of HGV movements results in a **negligible increase** in the pollutant concentration experienced on average along three representative routes: an increase in nitrogen dioxide (NO_2) and particulate matter (PM_{10}) of less than 0.25% of the respective air quality annual objective (as set out in Table 1).
- Despite contributing a negligible amount to total pollutant concentrations, emissions from a rail freight service generally affect more people than emissions from the equivalent road freight journeys. This is primarily a consequence of two factors: rail routes passing through areas of higher population density and pollutants emitted by rail being released at a higher speed and temperature compared with road emissions, resulting in them being released higher into the atmosphere and being spread over a wider area.
- Compared with road, particulate emissions improve by a factor of 2 when the Class 70 locomotive is considered rather than the oldest Class 66s.

Exhaust characteristics (velocity and temperature) are crucial. When these are taken into account, dispersion modelling shows that the emissions plume reaches around three times the height of the locomotive. Hence, near the track, there is less impact at ground level from rail emissions compared with road emissions. However, although concentrations then fall off away from the rail line, at further distances away, the change in concentrations is larger than for road. Simplistically, road emissions can be considered to act like a 'lighthouse', emitted radially around the source. Rail emissions are more like an 'anti-aircraft battery', a high-velocity narrow stream ejecting to a much higher level in the atmosphere before falling back to ground level further away.

Increasing pollutant concentrations by a small amount over a wide area might not be a cause for concern for pollutants like NO_2, where policy focuses on reducing concentrations at hotspots exceeding this pollutant objective. On the other hand, the UK government has recently introduced targets for reducing exposure to particulates ($PM_{2.5}$) of the overall population across England, aiming to reduce small but diffuse contributions, including those from rail freight.

When looking at population exposure (combining increases in pollution with population density) along the three representative flows considered, rail freight performed generally better than road freight on inner London routes, having smaller impacts on both NO_2 and PM_{10} exposure. Road freight performed markedly better than rail freight in situations where HGVs travel mostly on motorways outside urban centres, while freight trains pass through densely populated areas.

In other situations, the comparison is more nuanced. The two modes either have similar impacts, or rail freight has larger impacts on population exposure to NO_2, while road freight has larger impacts on exposure to PM_{10}. The exact building configuration and population distribution would need to be considered location by location to establish exactly how each mode compares along a specific route.

Recommendations for future work that have emerged from this pilot study include:

- using more complex building configurations with a realistic mix of different building types
- refining the population distributions, particularly for apartment buildings
- developing a better understanding of abrasive emissions and $PM_{2.5}$ emission factors
- comparing modelling approaches against monitored data.

Although rail emissions do not affect most buildings immediately next to the railway, they can affect higher floors of blocks of flats due to the barrier effect of the buildings limiting pollutant dispersion. As a result, more refined modelling of the circulation of pollutants around such buildings is needed. There is little information publicly available on abrasive emissions from trains. These are likely to be localised where trains brake or the track curves, and incorporating this spatial variation would give a more accurate picture.

Abbreviations

Abbreviation	Definition
ADMS	Atmospheric Dispersion Modelling System
AQO	Air quality objective
CERC	Cambridge Environmental Research Consultants
CFD	Computational fluid dynamics
CO_2	Carbon dioxide
DfT	Department for Transport
EFT	Emission Factor Toolkit
GB	Great Britain
GHG	Greenhouse gas
GIS	Geographic information system
GOBLIN	Gospel Oak – Barking Line
HGV	Heavy goods vehicle
HVO	Hydrotreated vegetable oil
IAQM	Institute of Air Quality Management
LAEI	London Atmospheric Emissions Inventory
LoHAM	London Highway Assignment Model
NAEI	National Atmospheric Emissions Inventory
NO	Nitric oxide
NO_2	Nitrogen dioxide
NO_x	Nitrogen oxides
O_3	Ozone
OHLE	Overhead line equipment
ONS	Office for National Statistics
PM	Particulate matter, includes PM_{10} and $PM_{2.5}$
PM_{10}	Particulate matter less than 10 micrometres in diameter
$PM_{2.5}$	Particulate matter less than 2.5 micrometres in diameter
REEC	Railfreight Energy & Emissions Calculator
RSSB	Rail Safety and Standards Board
SEEsim	Speed, Energy & Emissions Simulator
UIC	International Union of Railways
UPRN	Unique Property Reference Number
WCML	West Coast Main Line
TAG	Department for Transport, Transport Analysis Guidance

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1 Introduction

Due to lower rolling resistance and higher energy efficiency, rail freight has clear benefits over road freight in terms of economic efficiency and in reducing greenhouse gas (GHG) emissions, road congestion, and road accidents. However, the urban air quality impacts of rail freight compared with road freight are poorly understood, and these could either support or potentially complicate the encouragement of modal shift.

Unlike emissions of carbon dioxide (CO_2), which have a global impact on climate, the health and environmental effects of emissions of air pollutants, such as nitrogen oxides (NOx) from combustion and particulate matter (PM) from combustion and abrasion, are more localised. Of most concern are situations where the ambient concentrations of key air pollutants, such as nitrogen dioxide (NO_2) and particulate matter less than 2.5 microns in diameter ($PM_{2.5}$), are close to or exceed current regulatory standards. The current understanding by medical science is that exposure to particulates should be reduced as much as possible, and a safe level of exposure cannot be defined, unlike for other pollutants.[1] For this reason, the UK government has recently set legally binding targets to reduce the exposure to $PM_{2.5}$ for the whole population of England[2] and introduced stricter standards to be achieved in interim stages between 2028 and 2040.

The current air quality legal framework drives central and local government regulatory efforts and initiatives, as well as public awareness and perceptions of public health issues. Until recently, exceedances of the NO_2 standards in urban areas were considered the biggest air quality challenge. The new $PM_{2.5}$ standard and emission reduction targets reflect how attention is now being focused on the marginal impact of all emission sources of both pollutants.

Rail emissions (especially of NOx) are increasing in relative importance as the road fleet is renewed and modernised faster than the rail fleet. Additionally, low emission zones (of varying scope) have been or are being evaluated and implemented. This may lead to local authorities, under public pressure, to restrict existing rail freight usage and operations, as well as restrict the development of new flows and terminals. A more detailed understanding of rail freight's impact on urban air quality, especially compared with road freight, is therefore needed to contribute to informed discussions, planning, and policy making, as well as to realistically support and communicate the rail freight industry's environmental benefits and societal contributions.

A previous assessment of rail freight emissions carried out for Rail Partners[3] by Aether, Railfreight Consulting, and the University of Hull developed average gram-per-tonne-kilometre factors for CO_2, NOx, and PM for road and rail (accounting for outbound/return loadings and 'First/Last Mile' legs). These estimates were based on 17 representative freight flows and their proportion of national traffic.

[1] https://www.gov.uk/government/collections/comeap-reports

[2] The Environmental Targets (Fine Particulate Matter) (England) Regulations 2023 (SI 2023 No.96).

[3] Gibbs, M. et al. (2023). Research on Environmental Performance and Future Outlook for Rail Freight, Report to Rail Partners (unpublished).

Although a 76% lower CO_2 figure for rail was demonstrated in the Rail Partners work, national average factors for NOx and PM emissions (on a gram-per-tonne-kilometre basis) were similar or higher than for road, driven notably by the early Class 66s locomotive, which only meet UIC1 emission standards. However, an inner-urban case study from Acton to Purley (and including associated heavy goods vehicle [HGV] local 'Last Mile' distribution to an example building site in South London) showed that specific rail emissions along this route were generally better than road, even for the older Class 66 UIC1 locomotive. For the Class 70 locomotive (which meets the more recent and more stringent Euro Rail Stage IIIA emission standard), rail emissions were as low as 24% of the road emissions. These results point to the case for improved locomotive combustion emissions performance or utilising electric traction wherever possible, but they also show the importance of rail having the potential to maintain a higher average speed in urban areas.

In reality, the air quality impact of rail may be better or worse than that of road in urban areas when considering how the emissions of pollutants are dispersed and the actual concentrations of those pollutants in the air that people breathe in typical and reasonable circumstances.

Road and rail freight emissions will have different profiles and different pathways to receptors. Consequently, the incremental increase in emissions and resulting concentrations from transporting the same amount of goods can be expected to vary by mode, and so will the population exposure vary along a particular route.

When discussing the impacts of air pollution on human health, the term 'receptor' is used to refer to locations where people are regularly present for periods of time relevant to the applicable air quality objective, and where pollutant concentrations in the outdoor air may affect what they breathe. Where relevant, the height of a receptor is assumed to be 1.5 m above ground level, colloquially called 'nose height'.

Detailed air dispersion modelling carried out by Aether for the area around Leamington Spa station in Work Package 3 of the RSSB T1186 project[4] showed that the release height of road vehicle exhaust emissions (typically near to the ground) and rail emissions (at rolling stock roof level) and the proximity of receptors to these sources (pavements and building frontages) have significant impacts on the concentrations experienced. Further research on these factors involving detailed air dispersion and exposure modelling could demonstrate rail freight's effective urban air quality impact.

The goal of this project was to provide a robust, evidence-based comparison of the local air quality impacts of rail and road freight movements within representative areas of Great Britain utilising available rail emissions and air dispersion modelling. The project aimed to:

- Revisit and confirm the findings of the previous Rail Partners report regarding total emissions from rail and road freight operations for representative freight flows.

[4] Brook, R. (2021). The Impact of Rail Emissions in the Leamington Spa AQMA. RSSB T1186 WP3 Report (unpublished).

- Quantify the local air quality impacts of three representative freight flows by determining changes in NO_2 and particulate matter concentrations, and evaluating the impact of key factors (e.g., height, distance) that affect concentrations experienced at different locations near a road or railway line.
- Generate rational and defensible evidence to support future modal shift policies and investment decisions in rail freight.

Key questions that this study was intended to address are:

- How much do rail freight emissions contribute to the concentrations of air pollutants that people are exposed to in typical cases?
    - Compared with overall existing concentrations?
    - Compared with the same amount of freight carried by road?
- What are the most important factors that make a difference to the concentrations of air pollutants from road and rail freight that people are exposed to in typical cases?
- How many people live close to typical rail freight and road freight routes? What additional concentrations will they be exposed to from rail freight?

This pilot study provides a foundation for future work by examining representative freight routes and the most widely used rolling stock in current service. Emissions from freight trains running in normal service, and their long-term impact on nearby populations, were considered. Emissions from stationary trains, which can lead to short-term local impacts, were not considered here. Further aspects for investigation are covered in the recommendations in the final section.

The report has been prepared and written to be accessible to a wide audience, including rail industry sustainability staff, policy makers, decision makers, and technical practitioners. Technical terminology is used where necessary and accompanied by concise explanations to aid interpretation. Where technical concepts are introduced, they are explained to provide sufficient context for interpreting the results without requiring specialist knowledge of air quality.

The rest of this report is structured as follows:

- Section 2 discusses factors that affect air quality and key concepts in this field.
- Section 3 describes the methodology for the whole-route emissions modelling, dispersion modelling, and population exposure modelling.
- Section 4 contains results and discussion.
- Section 5 contains key learnings and recommendations.
- The appendices contain supporting information for the emissions, dispersion, and exposure modelling.

2 Factors affecting air quality

2.1 Overview

Decarbonisation is a global issue; exactly where CO_2 emissions occur is not significant. Thus, to address global warming, simple comparisons can be made, such as rail produces 76% less CO_2 than road.[5] However, air quality is inherently a local issue since concentrations of the main air pollutants, such as NOx and PM, will vary significantly spatially, as will their impact on human health and the environment at different locations. Two questions need to be considered:

- What is the nature of the air pollutant emissions (where, how, and in what quantities are they are released)?
- How do those emissions disperse from the point of release and translate into ambient pollutant concentrations at locations where people and sensitive environments may be exposed?

2.2 Air pollutant emissions and ambient concentrations

Air pollutant emissions and ambient pollutant concentrations are distinct concepts. Emissions are the quantity of pollutants directly released from a source, whereas concentrations are the amounts of pollutant present in the air at a given location after dispersion and other atmospheric processes (such as chemical transformation). Once released, pollutants may disperse (i.e., spread out or become diluted) or accumulate within a particular location (depending on the local topography or building configurations), and they may undergo chemical transformation within the atmosphere, all of which influence the concentrations experienced at a receptor (i.e., where people breathe). It is these ambient concentrations, rather than emissions, that determine human exposure and therefore form the basis of air quality standards and health impact assessments.

2.3 Standards for pollutant concentrations and impact magnitude

Poor air quality is linked to 30,000 deaths per year in the UK.[6] NOx forms during fuel combustion, while PM is released both during fuel combustion and from abrasion (e.g., of rubber on road or steel wheel on rails). Reducing the public's exposure to these pollutants is therefore a key environmental and public health priority.

Concentrations of air pollutants are typically measured as $\mu g/m^3$ and are averaged over a particular timeframe (e.g., hourly or annual averaging periods) for comparison against a particular air quality objective or target.

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[5] Rail Partners. (2023). Freight Expectations.

[6] Royal College of Physicians. (2025). A breath of fresh air: responding to the health challenges of modern air pollution.

Objectives for NO_2, PM_{10}, and $PM_{2.5}$ concentrations were set by the UK Air Quality Strategy,[7] Air Quality Standards Regulations 2010 (amended 2020),[8] and the Environmental Targets Regulations 2023.[9] These objectives, as detailed in Table 1, apply outside of buildings (i.e., façades) where members of the public may be regularly present over the averaging period. Such locations include building façades of homes, schools, hospitals, care homes, and public buildings, while the objectives do not generally apply at locations where the public would not be expected to have regular access (e.g., kerbside of motorways, railway embankments). This study therefore focuses on receptors relevant to the annual mean objectives, represented by the façades of buildings along the assessed routes, which, for the purposes of this assessment, have been assumed to be residential properties.

Table 1 UK annual mean air quality objectives for NO_2 and PM_{10} and target for $PM_{2.5}$

Pollutant	Averaging period	Concentration	Exceedance allowance
NO_2	Annual mean	40 µg/m³	Not to be exceeded in a year
PM_{10}	Annual mean	40 µg/m³	Not to be exceeded in a year
$PM_{2.5}$	Annual mean	20 µg/m³	Not to be exceeded in a year
		12 µg/m³	Interim target to be achieved by 2028
		10 µg/m³	To be met across England by 2030 (target) and 2040 (statutory)

The scale of impact magnitude has been defined in line with the Institute of Air Quality Management's (IAQM) guidance impact magnitude criteria,[10] as detailed in Table 10 in Appendix B.

[7] Department of the Environment, Transport and the Regions in Partnership with the Welsh Office, Scottish Office and Department of the Environment for Northern Ireland. (2007). The Air Quality Strategy for England, Scotland, Wales, Northern Ireland. HMSO, London.

[8] The Environment (Miscellaneous Amendments) (EU Exit) Regulations 2020 (SI 2020 No.1313), which amend the Air Quality Regulations (SI 2010 No.1001) to account for EU withdrawal.

[9] The Environmental Targets (Fine Particulate Matter) (England) Regulations 2023 (SI 2023 No.96).

[10] IAQM. (2017). Land-use Planning & Development Control: Planning for Air Quality. V2.2. Institute of Air Quality Management, London. http://iaqm.co.uk/guidance/

2.4 How emissions and concentrations can vary

Multiple factors influence the ambient air pollutant concentrations to which the local population is exposed. For all diesel-powered vehicles (locomotives or road vehicles), the rate at which air pollutants are emitted is primarily a function of the engine technology and its operating condition. Emissions vary continuously according to the power demand placed on the engine, which is influenced by the characteristics of the vehicle, its load, speed, and acceleration, and the operating environment.

Once emissions are released, the concentrations experienced at a particular receptor nearby will depend on how pollutants disperse and transform in the atmosphere. Factors influencing ambient pollutant concentrations include:

- wind speed and direction (which determine where pollutants travel and how quickly they disperse)
- atmospheric stability (how well the atmosphere mixes pollutants)
- vertical and horizontal separation between the emission source and receptor (the distance and height between where pollutants are released and where people are exposed)
- effects of buildings, embankments, and other structures effects (including shielding, which can block, channel, or trap pollutant dispersion)
- topography (the shape of the surrounding landscape, such as hills or valleys)
- existing ambient pollutant concentrations (the pollution already present in the air from all existing sources before considering the freight movements being assessed)
- chemical transformation (reactions in the atmosphere that can change one pollutant into another, such as NOx being converted to NO_2)

Although this study considers routes with both urban and rural components, densely populated urban areas are likely to experience the greatest public health impacts because larger numbers of people are exposed to changes in pollutant concentrations. Furthermore, background air quality (which is influenced by other sources such as road traffic and domestic combustion) is generally the poorest in densely populated urban areas. As a result, potential increases in pollutant concentrations from rail freight operations (or the equivalent road traffic) are likely to be of greatest significance in densely populated urban areas, where population exposure and existing background concentrations are highest.

3 Methodology

3.1 Route emissions modelling

3.1.1 Route selection

Three suitable representative rail freight corridors and corresponding road routes, incorporating 'First and Last Mile' connections, were selected to maximise the potential for transferrable learnings across different operational and geographic contexts. These are:

- Tunstead – Battersea (aggregates): one rail service equivalent to 91 HGVs
- Merehead – Acton – Bow (aggregates): one rail service equivalent to 76 HGVs
- London Gateway – Middlemore Industrial Estate, West Midlands (intermodal): three services per day, each equivalent to 50 HGVs.

Maps of these routes are shown in Figure 1 and Figure 2. More details on these routes and the rationale for their selection are provided in Appendix A.

The most efficient routing for road and rail was considered for each flow. In some cases, the relevant road and rail corridors were generally adjacent, whereas in other situations, the relevant road and rail routes were quite separated due to nature of the rail and major road networks.

Although these freight flows all include rural components, the focus was on flows transporting goods for distribution and use within urban areas, where both road and rail must service customer demand within densely populated zones, which are also areas where air quality is generally poorest, as explained in Section 2.4. Rail routes within London were considered since they pass through a large number of densely populated and built-up areas, enabling a good understanding of impacts on air quality in urban areas. The availability of the comprehensive and high-quality dataset from the London Atmospheric Emissions Inventory (LAEI) also supported the selection of the London routes and the assessments carried out in this study.

Figure 1 Representative routes considered in this study

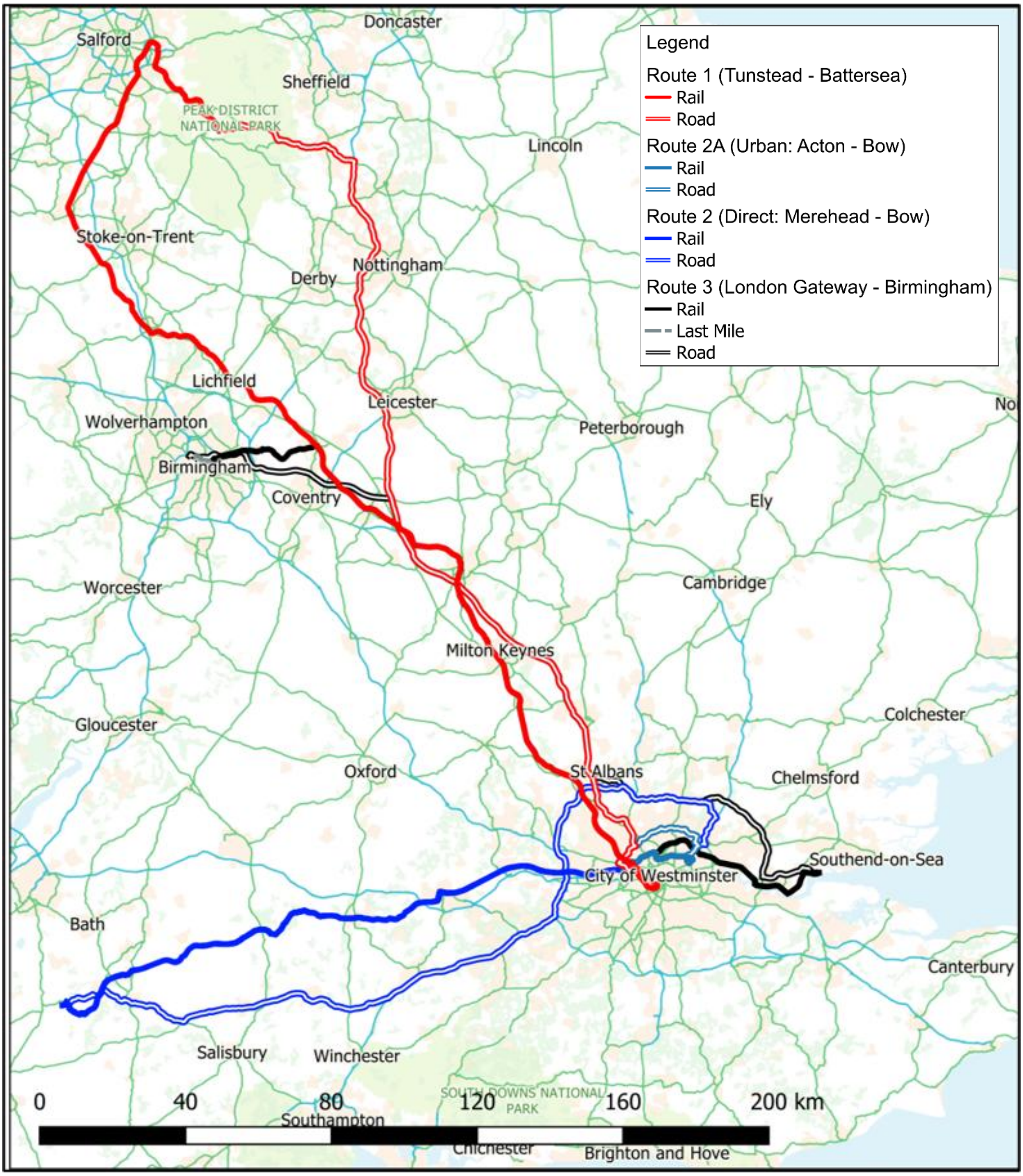
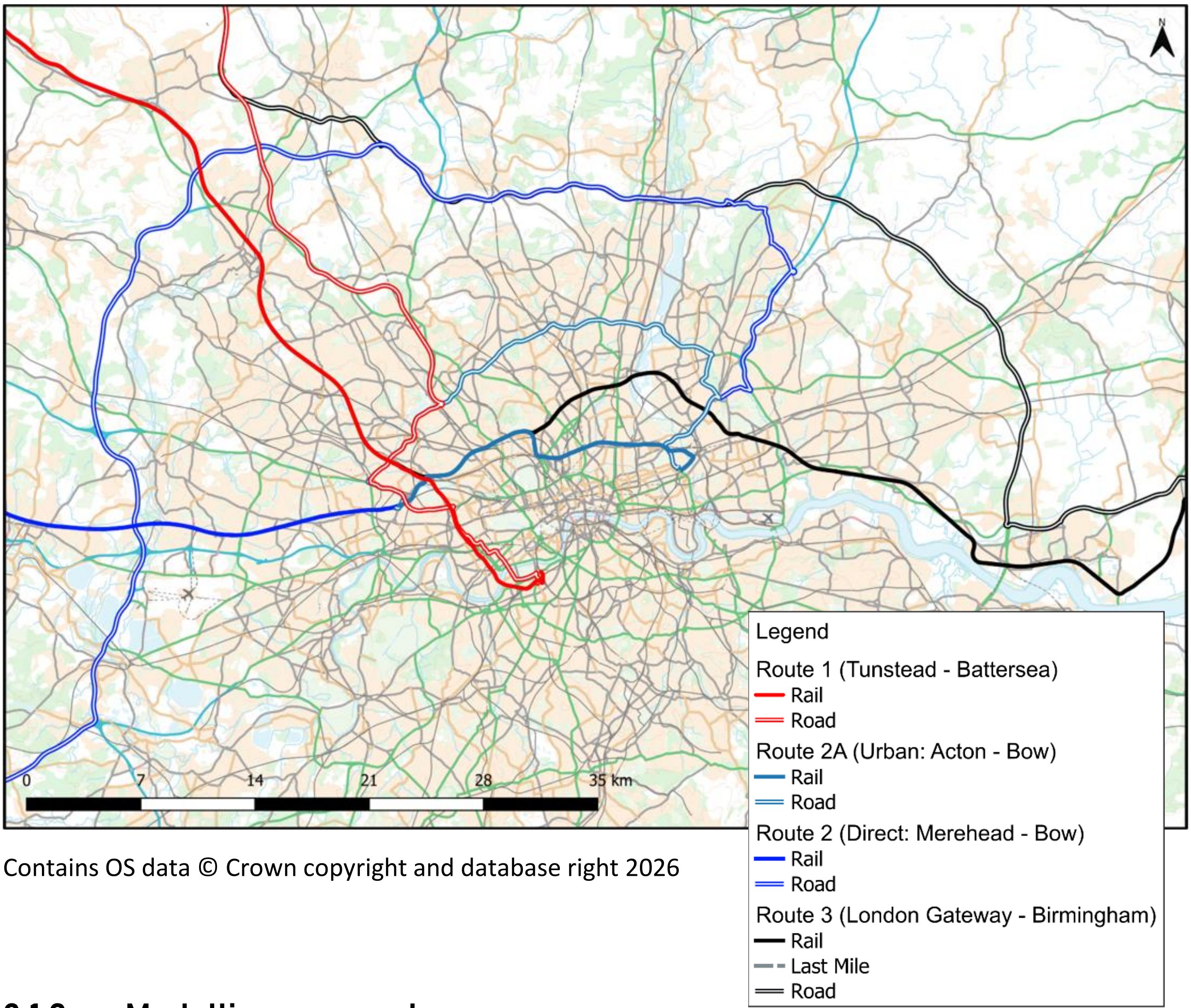


Figure 2 Representative routes considered in this study (London area)



3.1.2 Modelling approach

The approach used in the earlier Rail Partners study was followed. Emissions from typical outbound and return rail flows (including 'First and Last Mile' HGV legs to/from the railhead where applicable) were first modelled.

As with the previous Rail Partners work, the Railfreight Energy & Emissions Calculator (REEC), now referred to as the Speed, Energy & Emissions Simulator (SEEsim), was used for the detailed modelling of rail emissions. SEEsim is a sophisticated journey simulator that can calculate train performance and timings as well as energy usage and emissions as the train accelerates, cruises, coasts, brakes, and idles while stationary. It was developed by the University of Hull and Aether and is now developed, maintained, and commercialised by Railwhere, a company that has been spun out from the University of Hull. SEEsim can represent the key characteristics of a range of locomotive and wagon types, and it can simulate running to schedules along any route on the GB rail network. Since SEEsim determines the required tractive effort and engine notch at any point along the route, the notch-based combustion emission factors for NOx, PM_{10}, and $PM_{2.5}$ developed in the

RSSB T1187 emission factors project[11] can be used to accurately determine the spatial variation in emissions along a route at 10-m intervals for a particular route gradient, line speed, locomotive type, and the number and type of wagons and their loadings.

For modelling both direct route-level and 'First/Last Mile' HGV road emissions, the same approach used for the Rail Partners report was adopted. This is based on reasonable average emission factors for HGVs carrying the same type of freight on an optimal routing via major roads. The aggregates flows are assumed to be carried by 20-tonne (t) rigid HGVs, and the multimodal flow by 22-t articulated HGVs, noting that the penetration of electric HGVs is assumed to be currently insignificant. Road emissions were calculated using the National Atmosphere Emissions Inventory (NAEI) emission factors[12] for NOx and PM and DESNZ emission factors[13] for CO_2, assuming representative HGV types, fleet composition, and road types for each route.

Given that around three quarters of the current Class 66 fleet meet the UIC 1 emissions standard, this locomotive type was the main focus for emissions and dispersion modelling in this study (newer Class 66 locomotives meet later, more stringent emissions standards). Selected dispersion modelling simulations considered the sensitivity of results for the cleaner Class 70 locomotive. Applicable emission limit values for NOx and PM for Class 66 and Class 70 locomotives are shown in Table 2 below. (Note that in the emissions standards, PM refers to PM_{10}.)

Table 2 Rail emission standards in g/kWh

Applicable locomotive classes	Engine build year	Emissions standard	Test drive cycle	NOx	PM
66	2002 or earlier	UIC1	ISO 8178:F	12	0.54
66	2003–2008	UIC2	ISO 8178:F	9.9	0.2
66, 70	2009 or later	Euro IIIA	ISO 8178:F	7.4	0.2

[11] https://www.rssb.co.uk/research-catalogue/CatalogueItem/T1187

[12] https://naei.energysecurity.gov.uk/sites/default/files/2026-04/RoadtransportEFs_NAEI24_v1.xlsx

[13] https://www.gov.uk/government/publications/greenhouse-gas-reporting-conversion-factors-2025

3.2 Dispersion modelling

The aim of the dispersion modelling was to compare the air quality impacts of emissions from freight trains with those from equivalent HGV movements, and to examine how these impacts are influenced by factors such as building configuration and distance from the source. Pollutant concentrations were modelled in a set of idealised simulations, which are broadly representative of key rail and road routes characteristics within and outside London but are not a detailed representation of any specific location.

3.2.1 Model summary

Air dispersion models have been widely used to understand the air pollution impacts of a range of source types occurring in different areas. Dispersion modelling allows determination of how pollutant emissions result in pollutant concentrations and exposure and enables comparison against air quality standards. Such modelling is widely used to assess compliance with air quality objectives and policies at particular locations, to test scenarios for new developments, and to support cost-effective mitigation strategies.

For this study, the Atmospheric Dispersion Modelling System (ADMS-Roads)[14] was used. This is a comprehensive dispersion model used for investigating air pollution problems, and it allows the accurate modelling of road emissions and other sources. ADMS-Roads has been extensively validated against both field and laboratory data sets and used for many years in support of local air quality management. It provides a robust estimate of air pollutant concentrations due to the inclusion of factors such as vehicle-induced turbulence, 'street canyons' (where a road is flanked by tall, continuous buildings), complex terrain, etc., while taking into account important input data such as background pollutant concentrations, meteorological data, and traffic flows.

More details on the model, its limitations, and how it was set up for the simulations carried out in this project are provided in Appendix B.

3.2.2 Modelling approach

This study focused on modelling long-term impacts on annual average concentrations of NO_2 and PM, as these are the most relevant for human health at a population level. Dispersion of pollutants was modelled for each hour of the year using actual measured meteorological parameters then averaged. Due to the intermittent nature of train emissions in particular, short-term impacts are possible in specific locations and cases, but they have not been explored in this pilot study. They could be investigated further with dispersion modelling techniques, as is recommended in Section 5.

To ensure computational efficiency and consistency with the pilot nature of this project, detailed dispersion modelling of road and rail emissions along full route lengths was not carried out. Instead, separate simulations of representative geographies and topographies along the representative routes were carried out to determine the potential incremental impact on air quality from road or rail freight. The range of simulations,

[14] https://www.cerc.co.uk/environmental-software/ADMS-Roads-model.html

each of which captures a distinct combination of emissions characteristics, local geometries, dispersion behaviour, and receptor environments, is described in Appendix B.

The simulations were designed to explore the relative importance of different factors in affecting dispersion, as explained in detail in the following sub-sections. To understand how pollutant concentrations vary with distance from the rail or road source, each configuration was modelled over representative lengths parallel and perpendicular to the rail line or road, capturing prevailing meteorological conditions resulting in areas where pollutant concentrations are expected to be highest, in line with standard air quality assessment practices and guidelines.[15]

Rail freight emissions were modelled following the alignment of the track. Emissions were assigned at a representative release height above ground level, reflecting the typical exhaust position of diesel locomotives. SEEsim rail emissions outputs were converted into suitable emission rates for ADMS-Roads and distributed along each rail section. The intermittent nature of train operations was represented through time-varying emission profiles, which can characterise the frequency and duration of train movements.

HGV emissions for the equivalent road freight flow were modelled following road alignments. To simulate the effect of free-flowing traffic, it was assumed that emissions are released at ground level and are affected by the air turbulence caused by a constant flow of moving vehicles, typical of an urban dual carriageway. The typical variation in traffic levels by hour of day and day of the week was included in the model.

ADMS-Roads was used to calculate the contribution of rail and road sources to ambient concentrations of NOx and PM. These results were post-processed by adding existing ambient concentrations, estimated by regional or national scale modelling, and accounting for chemical reactions that convert NOx to NO_2 in the atmosphere. For further detail, see Appendix B.

3.2.3 Effect of distance, height, and buildings

The proximity of buildings to a source can greatly affect airflow and therefore the dispersion of pollutants. A particular case is when a road or rail line runs between two rows of tall buildings with little gaps between them, commonly called a ‘street canyon’. Several configurations of building heights and distances from the road or rail line were explored. Examples typical of urban areas included two-floor terrace houses, three- to 4-floor Victorian terrace houses or small blocks of flats, and seven- to 10-floor tall buildings, such as office buildings or large blocks of flats.

The release height of emissions is also an important factor because airflow changes with height. This was explored by simulating typical cases, such as viaducts and cuttings.

[15] Defra. (2022). Local Air Quality Management Technical Guidance (England) 2022 (TG22). HMSO. https://laqm.defra.gov.uk/air-quality/featured/uk-regions-exc-london-technical-guidance/

3.2.4 Effect of locomotive exhaust characteristics

In the first set of rail simulations, rail emissions were treated as road sources in ADMS-Roads, where, among other differences, the emissions are assumed to be released at ambient temperature and do not account for plume rise associated with exhaust temperature and exit velocity (which is the case for road vehicles). This enabled the use of the ADMS-Roads Canyons Module to investigate the effects of buildings on the dispersion of emissions. However, this approach does not represent specific rail emissions release characteristics (temperature and exit velocity). For instance, Class 66 emissions are ejected at 360°C and at 60 mph (27 metres/second) at Notch 8.

As a sensitivity test, consideration of rail emissions as a line source in ADMS-Roads was carried out, which enables understanding of the importance of more accurately representing the locomotive exhaust characteristics (i.e., temperature, exit velocity, and direction) on the dispersion of combustion emissions.

3.2.5 Effect of locomotive type

To understand the importance of locomotive type, a simulation run was carried out, specifically to compare the earliest Class 66s (which meet the UIC1 emissions standards) and the more recent Class 70 (which meets the newer Euro IIIa emissions standards). In the simulation, both locomotives were modelled as line sources, running at the same engine power (Notch 4) and at the same speed. However, emission parameters (NOx and PM_{10} emission rates, primary NO_2 fraction) and exhaust parameters (temperature and volumetric flow) were set at typical values for each locomotive. Details of these parameters can be found in Table 5 and Table 6 of Appendix B for the Class 66 and Class 70, respectively.

3.3 Route exposure modelling

The simulation results from dispersion modelling were combined with a GIS analysis of population density in the vicinity of the rail and road routes considered in Section 3.1 to estimate population exposure to the change in concentrations from the transport of road versus rail freight in urban areas.

From the modelling results of the simulations described in Section 3.2, a number of concentrations fall-off curves were derived, which are specific to the transport mode (road freight or rail freight) and the building configuration near the source (terrace houses, tall buildings, tall terraces). These curves were derived by modelling an emission rate of 1 g/km/s or 1 g/m/s (for road and line sources, respectively) and are independent of the emission rate at a specific source or location. The actual increase in concentration at a given distance from the source can then be calculated by multiplying the value of the curve by the emission rate at the source.[16]

Then, each of the three rail and road routes was divided into segments. The emission rates of NOx and PM_{10} from each segment (in g/km/s or g/m/s) were calculated based on speed and notch (for rail) or speed (for

[16] The approach of defining a curve of change in concentration per emission rate (expressed in µg/m^3 per g/km/s) is derived from the Design Manual for Roads and Bridges (DMRB) Screening Method, as described in Highways Agency, HA 207/07 Air Quality, May 2007.

road) and number of vehicles on the route. For each segment along a route, a building configuration from Section 3.2 was determined to be the most representative, and then the appropriate fall-off curve of pollutant concentrations was selected.

For road, the flat terrain (baseline) simulation was used for open countryside, parks, and large car parks but also areas of low-density or isolated housing near roads or where housing was over 25 m away from the road. There, terrace houses, blocks of flats, and large terraces (canyon) simulations were used where applicable. For this route exposure work, detailed consideration was not made of deep cuttings, tunnels, or viaducts.

For rail, the flat terrain line source simulation was used because this best captured the expected characteristics of locomotive plume. Since emissions are projected well above the locomotive, it was not considered necessary to consider nearby housing except for large blocks of flats. In that particular case, the simulation of rail as a road source that accounted for the effect of a tall building was used as a conservative approach.

The number of people living within five distance bands of each road and rail segment was then determined (see Appendix C for more details), and the average change in concentration at each distance band was calculated by multiplying the emission rate by the concentration fall-off curve as described above. The size of the population affected was represented using an impact score (change in concentration times population). Averaging along the full length of each route, the change in population-weighted exposure was derived for each train type and the equivalent number of HGVs.

4 Results and discussion

4.1 Whole-route emissions comparisons

Total emissions and emissions on a grams-per-tonne-kilometre basis (essentially for comparing emissions based on the same tonnage of goods moved over the same distance) for the three representative freight flows are shown in Table 3 below. Values are for the full routes, and these include the last-mile HGV leg for the rail flows where appropriate.

Table 3 Whole-route rail and road emissions

Route	Mode	Load (tonnes)	Distance (km)	Vehicle type	Total emissions (kg)				Emissions (g/tonne-km)			
					CO_2	NOx	PM_{10}	$PM_{2.5}$	CO_2	NOx	PM_{10}	$PM_{2.5}$
1. Tunstead – Battersea	Rail	1,824	348	Class 66 UIC1	8,028	138.70	4.30	3.20	12.6	0.219	0.007	0.005
				Class 66 Euro IIIA	8,027	64.70	1.25	0.94	12.6	0.102	0.002	0.001
				Class 70	7,310	57.70	0.40	0.30	11.5	0.091	0.001	0.000
	Road		283	91 x Rigid HGVs	8,960	33.76	5.33	3.01	95.0	0.065	0.010	0.006
2. Merehead – Bow	Rail	1,520	195	Class 66 UIC1	3,223	55.23	1.74	1.30	10.9	0.186	0.006	0.004
				Class 66 Euro IIIA	3,455	27.69	0.52	0.38	11.6	0.093	0.002	0.001
				Class 70	2,546	21.50	0.15	0.11	8.6	0.072	0.001	0.000
	Road		248	76 x Rigid HGVs	5,761	25.54	3.89	2.20	95.0	0.068	0.010	0.006
3. London Gateway – Middlemore Ind. Estate	Rail	891	255	Class 66 UIC1	10,734	164.16	5.03	4.73	47.2	0.722	0.022	0.021
				Class 66 Euro IIIA	10,765	74.16	1.58	2.14	47.3	0.326	0.007	0.009
				Class 70	10,528	75.56	0.60	1.40	46.3	0.332	0.003	0.006
	Road		238	50 x Articulated HGVs	25,720	6.24	2.65	1.54	121.2	0.029	0.012	0.007

A comparison of rail versus road emissions for the three freight flows is shown in Table 4 below. In each case, the rail emissions as a percentage of the road emissions are shown.

Table 4 Whole-route rail versus road emissions comparisons

Locomotive	CO_2			NOx			PM_{10}			$PM_{2.5}$		
	Tunstead – Battersea	Merehead – Bow	Gateway – Middlemore	Tunstead – Battersea	Merehead – Bow	Gateway – Middlemore	Tunstead – Battersea	Merehead – Bow	Gateway – Middlemore	Tunstead – Battersea	Merehead – Bow	Gateway – Middlemore
Class 66 UIC1	13%	11%	39%	334%	274%	2,454%	65%	57%	177%	85%	75%	287%
Class 66 Euro IIIA	13%	12%	39%	156%	137%	1,109%	19%	17%	56%	25%	22%	130%
Class 70 Euro IIIA	12%	9%	38%	139%	107%	1,130%	6%	5%	21%	8%	7%	85%

For CO_2, rail emissions are significantly lower, particularly for the aggregates flows. For PM, emissions are lower for the bulk flows but higher in some cases for the intermodal containers flow. There is a significant difference in emissions performance between the Class 66 UIC 1 and the Class 70 Euro IIIA (by a factor of 2 or larger), reflecting the tighter PM emissions standard for the newer Class 70 locomotive. For NOx, rail emissions are higher than road in all cases, reflecting the widespread introduction of Euro VI HGVs with significantly reduced NOx emissions compared with the earlier HGV emissions standards and the slower turnover of the rail locomotive fleet.

4.2 Dispersion modelling

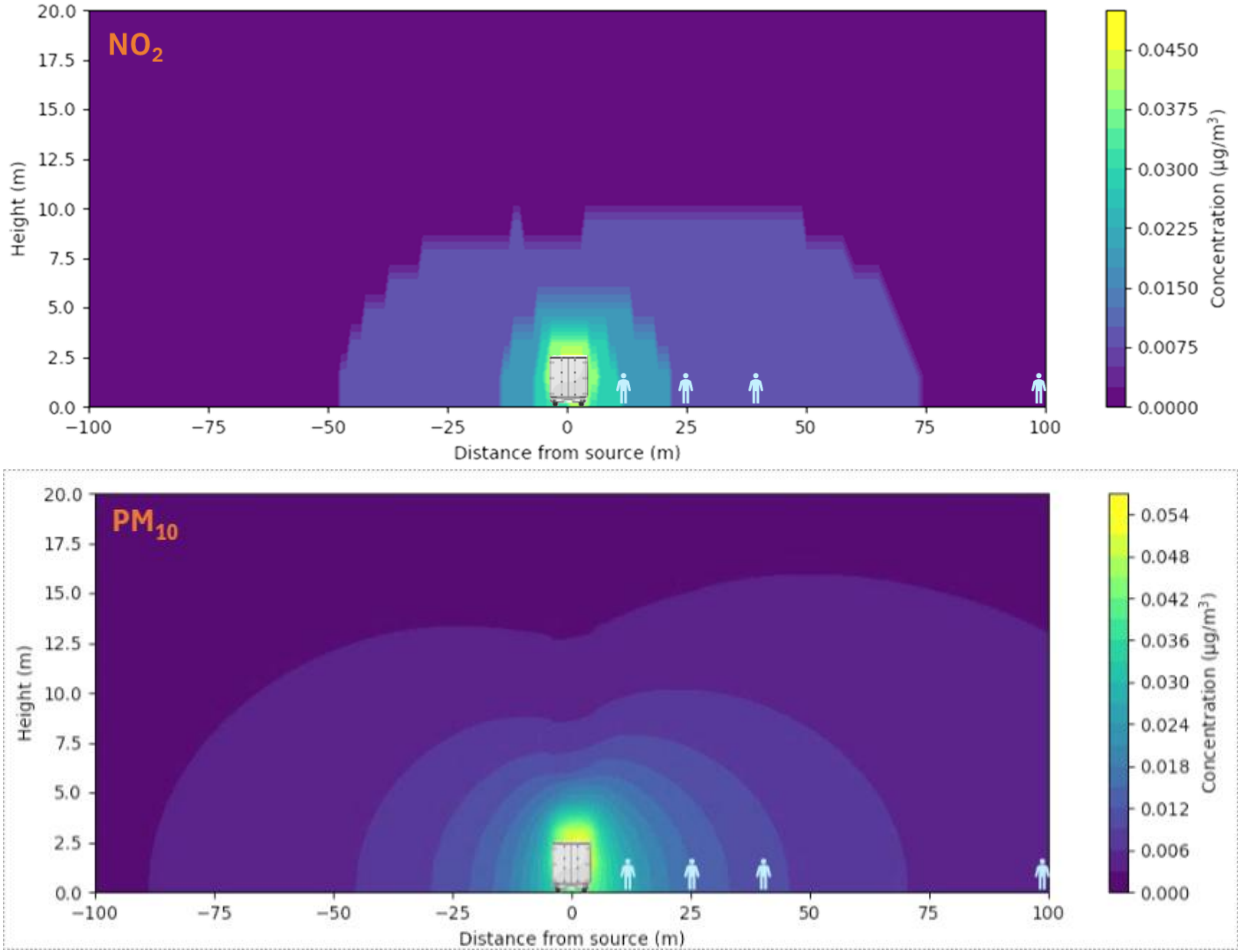
In this sub-section, results for road simulations are presented first, showing the effects of distance and buildings, then the results for rail simulations for these factors are presented.

4.2.1 Road freight

Figure 3 below shows a transect of concentrations of NO_2 and PM_{10} for the road base case (open terrain), which represents 91 HGV return trips in a day, the equivalent of one daily aggregates train. The prevailing wind direction is south westerly (as shown in Figure 14 in Appendix B, and which is from left to right in Figure 3), so concentrations are higher downwind at the same distance from the source. Nevertheless, some elevated concentrations are experienced upwind of the prevailing wind direction since the wind direction is not consistent all year.

The low resolution of the NO_2 contours compared with the PM plot is due to limitations in the tools used to calculate the conversion of NOx to NO_2 in the atmosphere (see Appendix B for further detail), which have a minimum resolution of 0.01 $\mu g/m^3$. Figure 3 shows the changes in NO_2 and PM_{10} concentration with distance away from the source. For both pollutants, the largest increase in concentration is closest to the source.

Figure 3 NO_2 and PM_{10} concentrations transect for road (no buildings)



Differences in NO_2 concentrations for road and different types of building configurations are shown in Figure 4 below. In the first plot (a road with terrace houses), most of the pollutants concentrations are trapped between the houses next to the road, resulting in concentrations that are three times higher than in the base case (open terrain with no buildings) at the same distance from the road. Dispersion beyond the terrace houses is same as the base case.

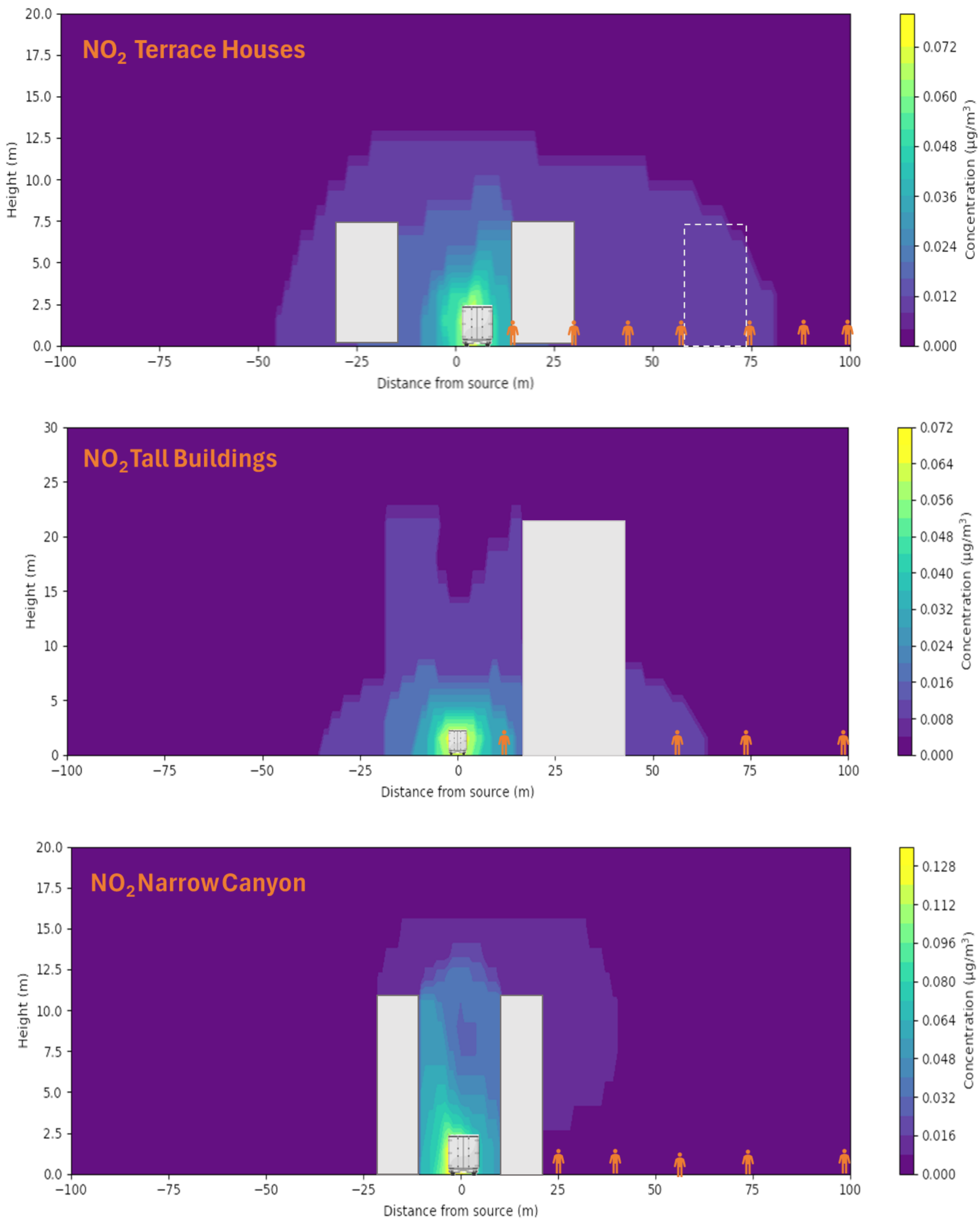
The first buildings next to the emission source were modelled explicitly as a street canyon. Other buildings farther away are not included in the model explicitly, instead being represented by dashed lines in the plot for reference. Other local urban characteristics, such as other buildings, were represented through the surface roughness parameter adopted in the dispersion model. Further details are provided in Appendix B.

For road with a tall building on one side (second plot in Figure 4), concentrations between the source and the building are higher than the base case, especially at higher floors, noting the street canyon effect limiting dispersion. There is some dispersion beyond the building, but concentrations are lower than the base case at the same distances from the road.

The plot also shows the complex effect that buildings have on airflow. The prevailing wind direction is from left to right, so the building is downwind of the road. However, the model shows increased concentrations also on the left (upwind) side of the road. This is because the building blocks the free flow of air and creates turbulence (eddies and vortices), which extend across the road. In the zone affected by turbulence, the vortices recirculate air and mix the pollutants emitted from the road throughout the zone. This turbulence zone is represented in the model with a box with sharp edges as a reasonable approximation.

The lowermost plot in Figure 4 is for road with tall terraces (i.e., a narrow street canyon), where emissions are highly concentrated between the buildings (nearly four times higher at the building façade compared with the base case. Dispersion beyond buildings is heavily blocked. This configuration is particularly relevant for modelling roads in densely built-up areas such as town high streets or 'mansion' terraces in Central London.

Figure 4 NO_2 concentrations transect for road (terrace houses, tall buildings, tall terrace houses)



4.2.2 Rail freight

Figure 5 shows the impact of distance and buildings on rail emissions. Here, the rail source is treated as a road source, i.e., the emissions are released at ambient temperatures from the realistic location on the top of the locomotive at 5 m height, which represents the total height above ground level (including the ballast, track, and height of the locomotive). This representation captures the effects of train-induced turbulence but does not account for plume rise associated with exhaust temperature and exit velocity, as detailed in Appendix B. Treating the rail emissions as a road source in ADMS-Roads allows consideration of the effects of buildings on shielding and reducing dispersion of pollutants farther afield from the track.

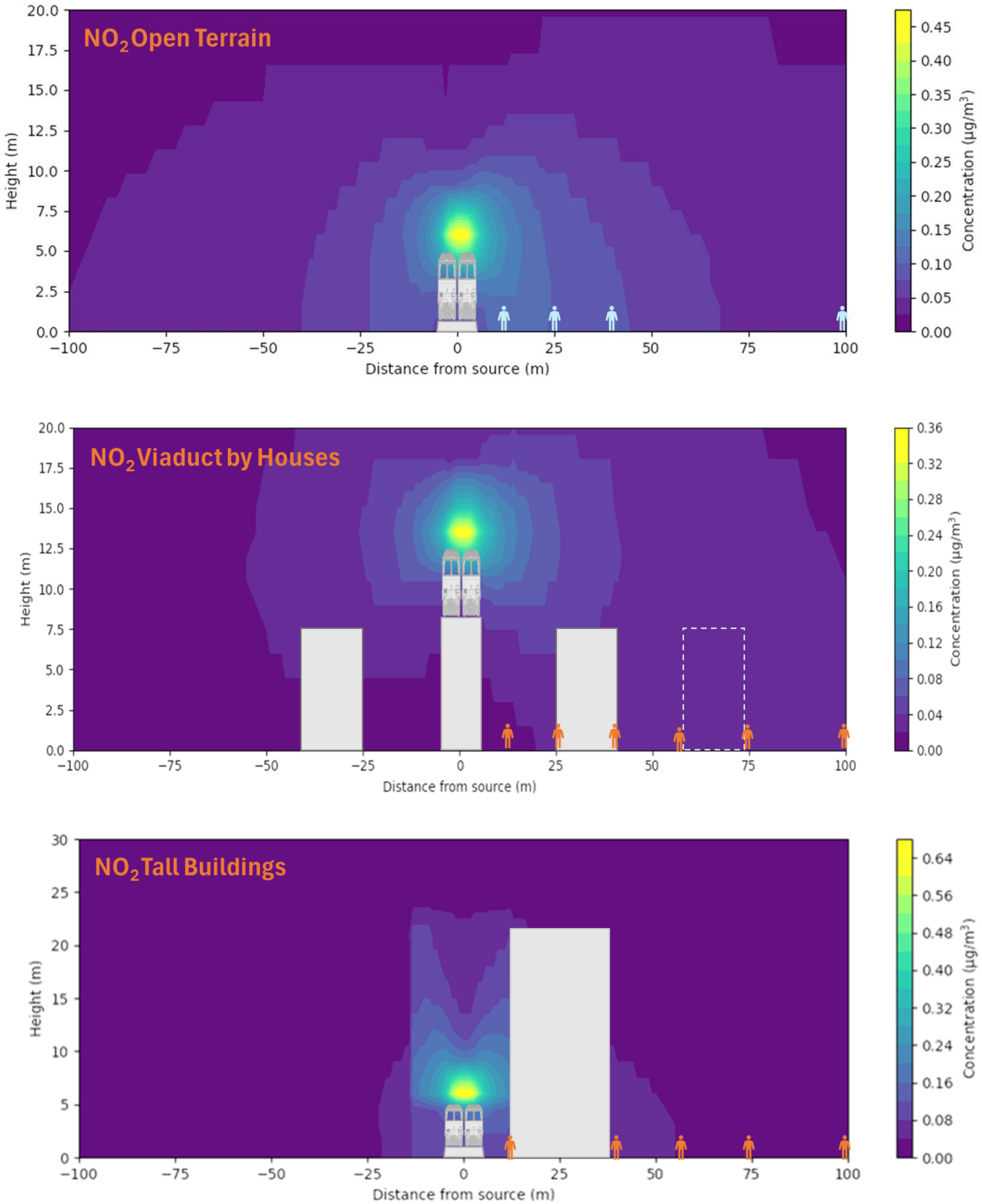
For the rail base case (first plot), concentrations at 1.5 m above ground level (representative of the average human breathing height) are highest within 25 m of the track and progressively declining farther away from the track.

The middle plot in Figure 5 shows the impact of a rail line on a viaduct. Wind speeds increase when the source is higher than ground level. Dispersion is therefore more effective, and concentrations below the viaduct at 1.5 m (i.e., breathing height) are lower than in the base case.

Similar to the roads set-up, the first buildings next to the emission source were modelled explicitly as a street canyon. Other buildings farther away are not included in the model, and these are represented by dashed lines in the plot for reference. Other local urban characteristics, such as other buildings, were represented through the surface roughness parameter adopted in the dispersion model. Further details are provided in Appendix B.

The lowermost plot in Figure 5 shows the impact of blocks of flats spaced regularly close to a rail line. As the exhaust plume is lower than the canyon, most of the concentrations are trapped between the source and the building. This shows that it is necessary to take into account higher concentrations at higher floors. There is some dispersion beyond the building, but concentrations are lower than the base case, i.e., the tall building provides a shielding effect.

Figure 5 NO_2 concentrations transect for rail (open terrain, viaduct, tall building)

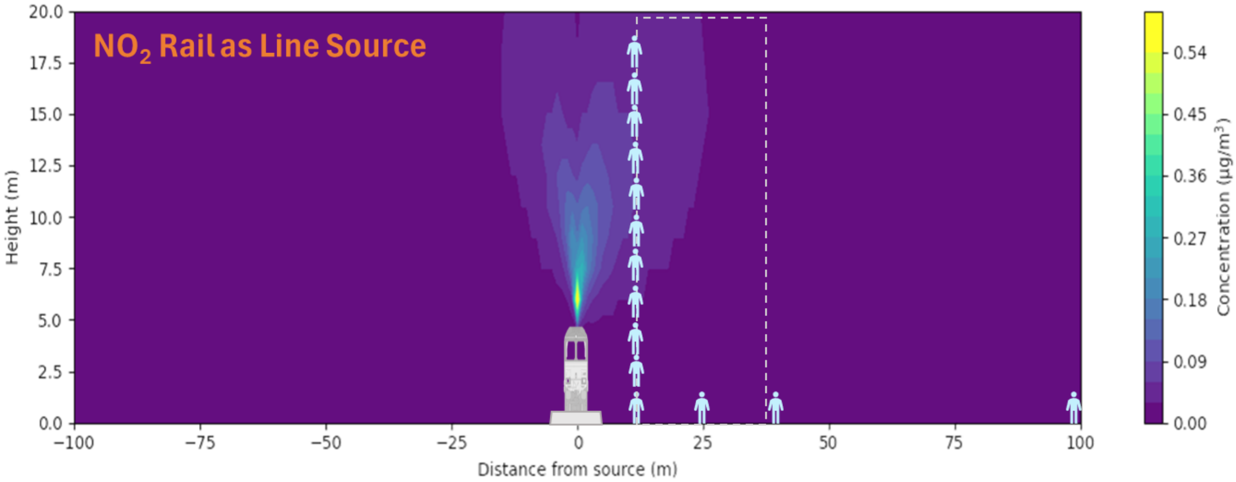


4.2.3 Effect of locomotive exhaust characteristics

In Figure 6 below, it can be seen that the plume modelled as a line source is more directed and rises much higher than in the base case, where rail freight is modelled as a road source. Consequently, there are much lower concentrations near the ground in the near vicinity of the railway line. Observations of plumes produced by locomotives late in the maintenance cycle, i.e., where the presence of lube oil helps to visibly delineate the plume, indicate a plume height that is around three time the height of the locomotive, consistent with these modelling results.

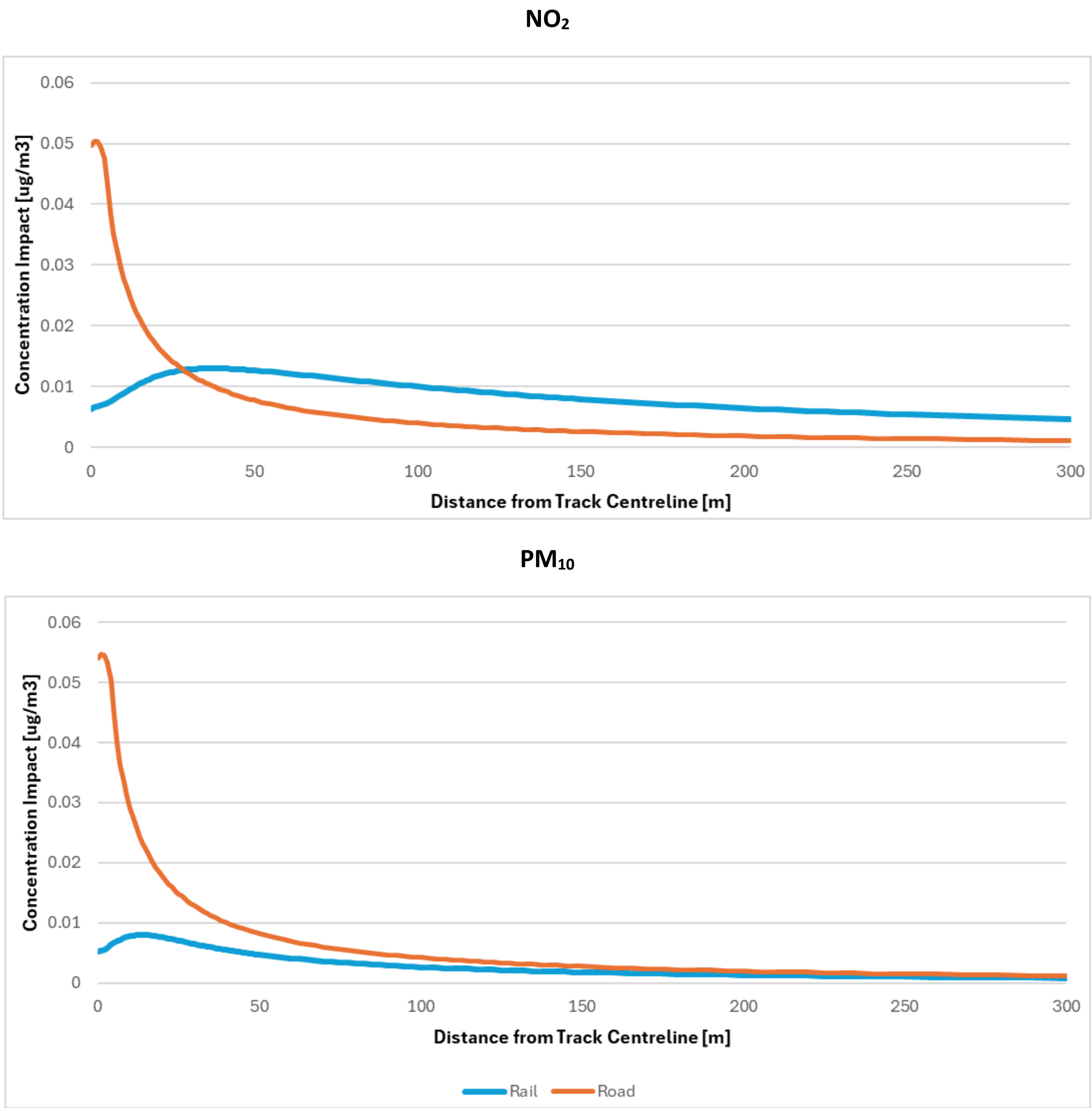
However, a key issue is that the ADMS-Roads does not allow line sources to interact with buildings. The outline of a tall building is shown in Figure 6, and it could be inferred that there would be an impact at the higher floors.

Figure 6 NO_2 concentrations transect for rail as line source of combustion emissions



The figure above shows how, when taking into account the effect of exhaust temperature and velocity on pollutant dispersion, the resulting distribution of concentrations is very different from the base case (Figure 5), where rail was modelled as a road source. However, importantly, farther away (beyond around 40 m) at nose height (1.5 m), NO_2 concentrations from rail freight are higher than for the equivalent road journeys when HGV emissions are treated as a road source, as can be seen in the fall-off curves for road and rail in Figure 7 below).

Figure 7 Comparison of modelled fall-off curves for NO_2 (top) and PM_{10} (bottom) concentrations away from track centreline for road emissions versus rail emissions (as a line source)



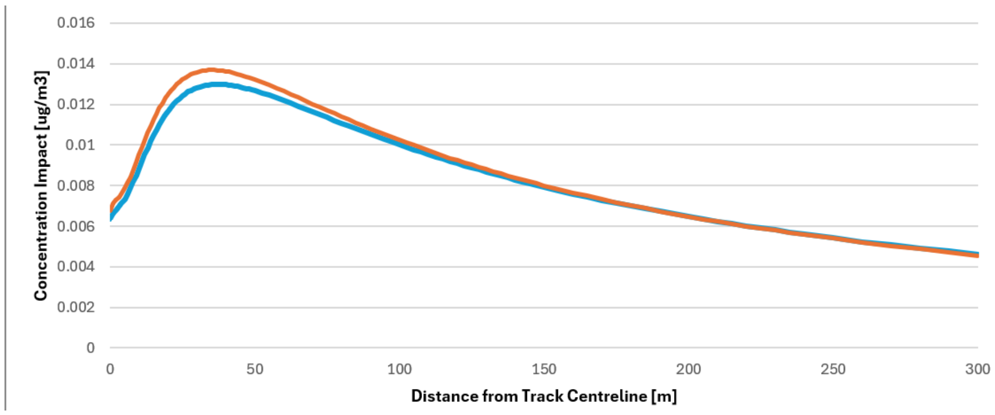
4.2.4 Effect of locomotive type

Figure 8 below shows the fall-off curves for concentration versus distance at a 1.5 m height (nose height) for Class 66 UIC 1 and Class 70 engines. These generally reflect the differences between compliance with the older and newer emission standards. At Notch 4, contributions to NO_2 concentrations from both locomotives are quite similar, as the Class 70 has higher primary NO_2 emission rates but also higher exhaust temperature and velocity, meaning the emissions are dispersed farther into the atmosphere. Contributions to PM_{10}

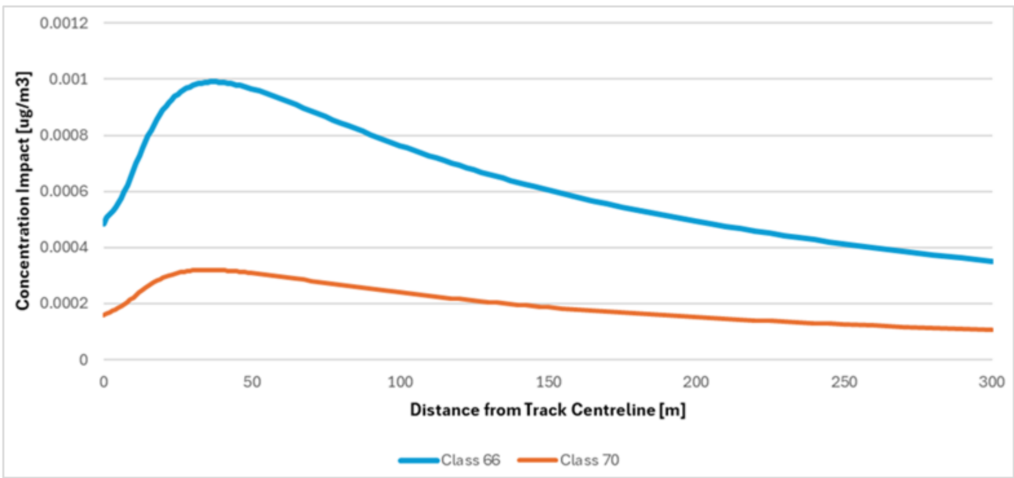
concentrations from the Class 70 locomotive are approximately 30% of the Class 66 contribution, in line with its lower emissions.

Figure 8 Modelled fall-off curves for NO_2 (top) and PM_{10} (bottom) concentrations away from track centreline for Class 66 UIC1 versus Class 70

NO_2



PM_{10}



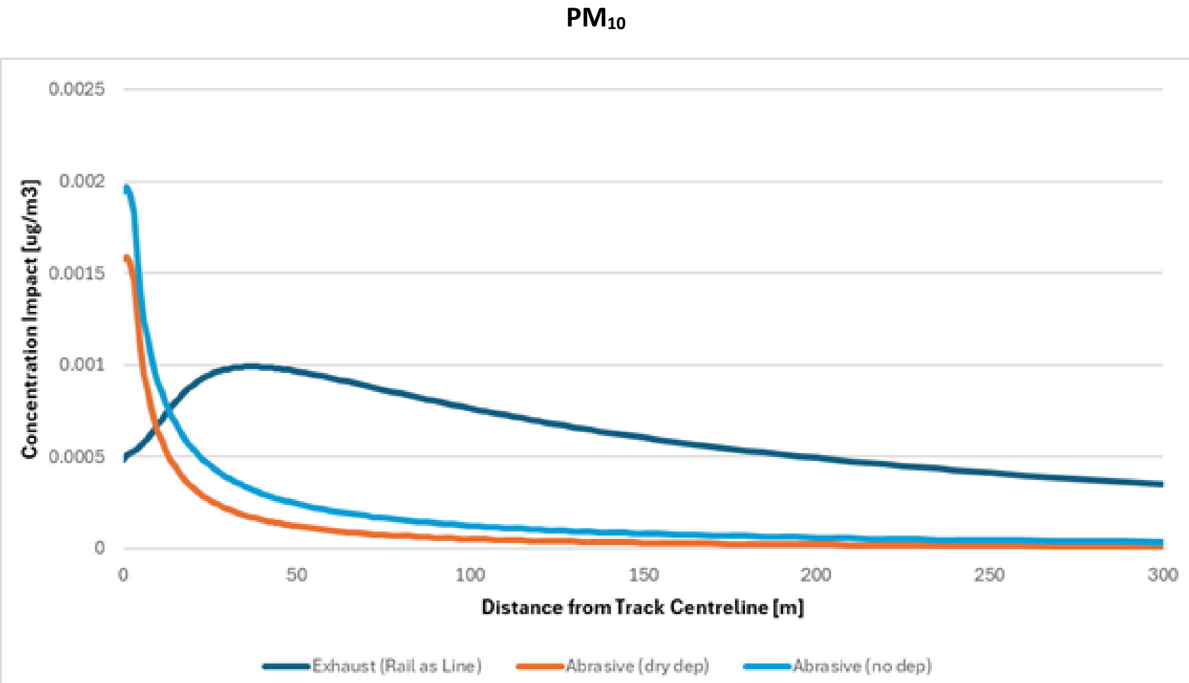
4.2.5 Rail abrasive emissions

Rail abrasive PM emissions arise from rail, wheel, and brake pad wear and, with more of a focus on combustion emissions, have been given limited consideration previously. Available suitable emission factors are limited. A factor of 0.013 g/vehicle-km has been developed based on Deutsche Bahn data[17] and validated

[17] https://iir.umweltbundesamt.de/2026/sector/energy/fuel_combustion/transport/railways/start

against UK estimates of rail, wheel, and brake pad wear[18], as detailed in Appendix B. (For electrified routes, abrasive wear of overhead line equipment [OHLE] contact wire and pantograph contact strips is around two orders of magnitude less than rail, wheel, and brake pad wear.) The distribution of such emissions will not be uniform along a route, being higher on curves and during braking. Furthermore, such emissions, being metallic in composition, will have a higher density than combustions emissions or tyre wear emissions and can be expected to settle out closer to the railway. This is shown in Figure 9 below, where modelled abrasive PM emissions fall out closer to the source compared with combustion PM emissions, which are dispersed farther, resulting in higher concentrations farther away from the source.

Figure 9 Modelled fall-off curves for combustion and abrasive rail PM_{10} emissions



[18] https://www.rssb.co.uk/research-catalogue/CatalogueItem/T1187

4.3 Route exposure modelling

The results of the analysis are shown in the charts in Figure 10 and Figure 11 on the following pages. Each chart shows the impacts from rail and the equivalent road journeys for the three selected routes, broken out by overall route, within London[19], outside London, and the 'Last Mile' (if applicable), for different distance bands along each of the three routes.

Figure 10 shows the increases in NO_2 concentrations. On average, both road and rail freight, in each distance band, result in small concentration increases (less than 0.1 µg/m^3), which are considered negligible when compared with the long-term standard (less than 0.25% of the annual NO_2 objective of 40 µg/m^3, in line with the IAQM guidance impact magnitude criteria[20]).

Road contributions to NO_2 concentrations (orange) are largest near the source (less than 25 m), more so within London than outside, due to the number of tall buildings and street canyons along the routes.

Rail contributions (blue) outside London are largest in the 25-m to 50-m band, due to higher exhaust speed and temperature pushing pollutants farther away from the source of emissions. Within London, this effect is modified by the effect of tall buildings, resulting in slightly higher increases in NO_2 concentrations between 0 m and 25 m away from the railway.

Rail contributions to NO_2 concentrations are larger than road on most routes and distance band, except within the 0-m to 25-m distance band, where road contributions are larger in most cases. An exception is Route 2A, where trains travel direct Acton to Bow on the North London Line and HGVs travel along the North Circular Road. On this route, road contributions are larger than rail up to 100 m away. Another exception is Route 3 within London, as emissions from HGVs on the M25 tend to be lower due to higher vehicle speeds, and rail emissions are larger since three trains are included in the flow.

Figure 11 shows the increases in PM_{10} concentrations. These are also small for both road and rail (less than 0.1 µg/m^3) and negligible if compared with the current long-term standard (less than 0.25% of the annual PM_{10} objective of 40 µg/m^3, in line with the IAQM guidance impact magnitude criteria).

On all routes considered, road contributions were larger than rail contributions to PM_{10} concentrations. This is because rail exhaust emissions of PM_{10} are lower than the equivalent fleet of HGVs for the same freight movement, and emissions are dispersed over a wider area, and particulates from rail abrasion are heavier than the same component from road vehicles (brake and tyre wear) and travel shorter distances. As for NO_2, road freight contributions are largest near the source and decrease as distance from the source increase.

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[19] The boundary between the 'within London' and 'outside London' zones is the outer edge of the M25 motorway.

[20] IAQM. (2017). Land-use Planning & Development Control: Planning for Air Quality. V2.2. Institute of Air Quality Management, London. http://iaqm.co.uk/guidance/

Figure 10 Freight NO_2 contributions along the study routes ($\mu g/m^3$) (rail = blue, road = orange)

Figure 11 Freight PM_{10} contributions along the study routes (µg/m[3]) (rail = blue, road = orange)

Figure 12 shows the NO_2 impact scores (NO_2 concentration increase times population). These are larger for rail (blue) than road (orange) in most cases, as the freight trains considered pass through the centres of towns and cities, where population is denser, while HGVs travel along motorways, which are typically built outside population centres.[21]

On inner London routes (Route 1 within London and Route 2A), where population density is similar for both modes, the impact scores of the two modes are comparable. Road freight scores are 35% larger than rail freight on Route 1 (London section) and 51% larger on Route 2A. On the other side, rail impacts are larger than road impacts at distances beyond 50 m from the track. A similar effect is seen on Route 3, where road impacts are larger than rail in the Birmingham area (outside London chart), although this is compensated when the 'Last Mile' leg of the rail route is taken into account (overall route chart).

Figure 13 shows the PM_{10} impact scores (PM_{10} concentration increase times population). Road impacts are markedly larger than rail impacts on inner London routes (Route 1 within London and Route 2A), where the two modes travel through similar population densities. Rail impacts are markedly larger than road impacts on Route 3 in London, where freight trains travel across inner London, while HGVs travel along the M25. In other situations (outside London), the impacts of the two modes are similar.

[21] For a breakdown of population by distance for each route, see Figure 16 in Appendix C.

Figure 12 Freight NO_2 impact scores (concentration x population) along the study routes (rail = blue, road = orange)

Figure 13 Freight PM_{10} impact scores (concentration x population) along the study routes (rail = blue, road = orange)

5 Key learnings and recommendations

Dispersion modelling of pollutant emissions from rail has not been previously considered with this level of detail. This pilot study has provided a more complete picture of rail freight's impact on local air quality and a comparison with the equivalent road freight.

On a full-route basis, because of the energy efficiency of rail transport, CO_2 emissions are substantially lower per unit of freight transported by rail than by road. However, rail freight service has higher NOx emissions than for the equivalent HGV journeys transporting the same load. This difference has increased in recent years with the introduction of HGVs meeting the Euro VI emissions standard, which perform better, especially compared with Class 66 locomotives that meet the UIC 1 emissions standard, which comprise the majority of freight locomotives currently in use in the UK. Although the overall environmental case for rail freight remains strong, these changes have lessened the air quality benefits of rail freight. This could be addressed by utilising cleaner locomotives, such as the Class 70, using bi-mode locomotives, or introducing electrification schemes. Furthermore, as both modes ultimately substantially electrify in the future, rail will hold an advantage regarding emissions of abrasive PM, both for the amount released and how it is dispersed since metallic rail abrasive emissions settle out closer to the track than PM produced by brake and tyre wear from road vehicles.

This study has considered the dispersion of air pollutant emissions once they are released from the locomotive or HGV and how this affects the concentration experienced at receptors, i.e., the air people breathe in where they live. The results show that **the impact of a typical daily rail freight service, or the equivalent number of HGV movements, is only a small increase (less than 0.1 µg/m^3) in the total concentration experienced at any location along three representative routes**. The contribution from the freight flow resulting concentrations, when compared with the long-term air quality standards (less than 0.25% of the annual NO_2 or PM_{10} objective of 40 µg/m^3), can be considered negligible, in line with the Institute of Air Quality Management's guidance impact magnitude criteria.

Given this important conclusion, the number of people affected by this small increase in pollution from a typical freight flow can be compared for road versus rail. A rail freight service generally affects more people than the equivalent journeys by HGVs. Variation in the numbers of people affected is driven by three key factors:

- **Exhaust characteristics:** Although road vehicle exhaust is released at ambient temperatures and accumulates within the immediate vicinity of the road, locomotive exhaust is released at high speeds and temperatures. The exhaust plume from a locomotive reaches two to three times the height of a locomotive and has limited impact at ground level in the nearby vicinity of the rail line. However, although concentrations fall off with distance for both road and rail, the increase in NO_2 concentrations at distances farther away from the railway line (e.g., 100 m to 200 m) would be relatively larger than from road vehicles.
- **Building configuration:** Terrace houses or tall buildings can result in higher concentrations in the immediate vicinity of a road or rail line while resulting in lower concentrations further afield compared with open terrain. This effect is particularly exacerbated for street canyons, such as tall Victorian

terraces or business districts. For tall buildings, more people can be exposed on higher floors, particularly when a tall building is located next to railway line.

- **Population along a route:** Due to the historic development of the rail network, rail freight passes nearer more population (e.g., along the West Coast Main Line [WCML]), while road freight in non-urban areas passes along motorways and major roads, where there is less population within 200 m. This difference is very clear for the representative intermodal flow, where the road traffic passes through a much lower proportion of urban areas.

When looking at population exposure (combining increases in pollution with population density) along the three representative flows considered, road freight has a larger impact immediately next to the source, while rail affects a wider area.

On inner London routes, rail freight had generally smaller impacts than road freight on both NO_2 and PM_{10} exposure. Road freight performed markedly better than rail freight in situations where HGVs travel mostly on motorways outside urban centres, whereas trains tend to pass through densely populated areas.

In other situations, the two modes either have similar impacts or rail freight has larger impacts on population exposure to NO_2, while road freight has larger impacts on exposure to PM_{10}. The exact building configuration and population distribution would need to be considered location by location to establish exactly how each mode compares along a specific route.

Aspects of this work that could be refined in future studies include:

- more detailed modelling of a mixture of building configurations, accounting for different types of buildings and gaps at different distances from the road or rail line
- refined characterisation of rail emissions within ADMS, including treating a train as a series of point sources to better represent transient operating conditions, such as exhaust plume rise from trains accelerating or idling (e.g., puff releases)
- more detailed population distribution assessment, including distinguishing between industrial, commercial, and residential addresses, and a more refined estimation of population numbers at each residential address (e.g., single-family dwellings versus blocks of flats)
- utilisation of the latest population data, as there have been multiple apartment developments completed along the study routes since the latest census
- a more detailed understanding of apartment buildings and how these could be affected by nearby rail emissions, which would include understanding the distribution of occupants on different floors and which apartments face the railway, as well as consideration of existing air quality measures, such as sealed windows and the presence of balconies
- consideration of newer locomotive classes (e.g., Class 93, Class 99) and the use of biofuels such as hydrotreated vegetable oil (HVO), for which emissions test data by notch will be required

- developing rail abrasive PM emission factors that are speed-dependent and that can be applied to locations where a train will be braking
- developing detailed $PM_{2.5}$ emission factors from both locomotive exhaust and rail abrasion to enable more direct comparison against the latest targets for $PM_{2.5}$ concentrations in ambient air and reduction in population exposure
- evaluation and validation of different modelling approaches for train emissions against monitoring data, supported by collaboration with model developers, to improve the representation of rail sources, which would be relevant for both exhaust and abrasive PM emissions.

Although this work has focused on rail freight, aspects of the approach used here could be applied to passenger rail — for instance, how exhaust characteristics affect dispersion and concentrations could be considered for passenger trains, especially in enclosed stations, where there will be implications for potential mitigation measures, such as station ventilation.

Appendix A: Route emissions modelling supporting information

Route selection

Two aggregates flows and an intermodal flow were considered. The aggregates flows both start from a quarry and have final inner-city destinations (i.e., concrete plants at the rail terminal), whereas the intermodal flow starts from a port and finishes at an inland industrial estate, requiring a 'Last Mile' HGV leg from the nearest rail terminal. Routes serving London were considered since these provide a range of extended (and important) urban situations that are worth investigating and that will provide the highest number of learnings. Such specific opportunities are inherently limited for other cities. However, the variations in building configurations/density and traffic speed used in the detailed dispersion model simulations will be widely applicable to parts of many other routes across the road and rail networks throughout the rest of the country.

Side-by-side route comparisons (e.g., where a road route closely parallels a rail route) were not made, but rather, the most reasonable route between origin and destination that road and rail flows would separately follow was considered. So, the intermodal rail flow passes through multiple urban areas (a consequence of the historical legacy of the development of the rail network), which can be avoided by the equivalent road flow on the motorway network. This approach enables a realistic comparison of road and rail emissions per tonne-kilometre as well as an assessment of the exposure to the concentrations of air pollutants from the road and rail flows experienced by populations along each route.

Route 1: Tunstead – Battersea (one rail service equivalent to 91 HGVs)

This route provides an evaluation of a flow of aggregates used in building construction from a major quarry in the Peak District to central London. There is no 'First Mile' or 'Last Mile' for rail, since the origin is the rail-served quarry and the destination is the rail-served concrete batch plant in Battersea.

One daily return train per day of 2,400 tonnes (t) (24 wagons) was modelled, headcode 6051 outbound and headcode 6M51 return.

The road equivalent flow would be 91 return trips of a 20-t payload tipper lorry. The urban sections in London between the M25 and Battersea are somewhat similar for rail and road. The rail route includes the WCML and West London Line, while the most reasonable road route to avoid congestion includes the M1, North Circular Road, and A40, before passing through Kensington, Chelsea, and Battersea.

Route 2: Merehead – Acton – Bow (one rail service equivalent to 76 HGVs)

This route enables evaluation of the North London Line, which is critical for current aggregates and intermodal flows. Similar to Route 1, there is no 'First Mile' or 'Last Mile' for rail, since the origin is the rail-served quarry and the destination is the rail-served concrete batch plant in Bow.

The current practice of splitting double-headed jumbo trains from Merehead at Hanwell Bridge Loop before taking the North London Line to Bow was modelled, as was an idealised rail flow to Acton for onward delivery by road around the North Circular Road to Bow (Route 2A).

Headcodes 6A80 plus 6L83 were used for the comparator rail flow, which are 4,200-t and 2,000-t aggregates trains, respectively.

The comparator road vehicle was a 20-t payload tipper lorry, both operating the full direct route from Merehead to Bow and from Acton to Bow.

Route 3: London Gateway – Middlemore Industrial Estate (West Midlands) (one rail service equivalent to 50 HGVs)

Road movements for intermodal traffic from Felixstowe and London Gateway will avoid London. They are rural motorway-based (A14/M6/M1 and M25/M1/M6, respectively), whereas all of London Gateway's rail movements and a substantial proportion of Felixstowe's are via inner-urban rail routes through East and North London.

The urban sections for the modelled rail route are the Gospel Oak – Barking Line (GOBLIN) and the North London Line. The remainder of the route follows the WCML to Nuneaton and then to the Landor Street terminal (a good example of an 'inner-urban' intermodal terminal). A 'Last Mile' section by HGV from Landor Street to the Middlemore Industrial Estate in West Bromwich (a representative final destination) follows the most reasonable route.

The comparator rail flow was three daily return trains (based on headcodes 4M91 outbound and 4L51 return), each consisting of 12 FWA wagons and 21 FEA wagons travelling via the GOBLIN, North London Line, and WCML. The comparator road flow was for the equivalent number of trips operated by an articulated HGV with a 22-t container load, operating the full route from London Gateway to the Middlemore Industrial Estate via the M25, M1, M6, and A38(M).

Appendix B: Dispersion modelling supporting information

Modelling software

This project utilised the Atmospheric Dispersion Modelling System (ADMS-Roads), a sophisticated software suite used to model air quality impacts from industrial, urban, and road sources. It was developed by Cambridge Environmental Research Consultants (CERC).[22] ADMS-Roads is widely used in the UK for modelling dispersion of emissions from transport sources, including accounting for turbulence and street canyons. It is used by local authorities, consultants, and government for policy, planning, and permitting purposes, and it provides robust, validated predictions of pollutant concentrations. It is recommended by Defra as an effective, efficient tool that avoids the need to carry out more complex computational fluid dynamics (CFD) modelling for most local air quality management purposes.

Inputs to ADMS-Roads include emission rate, emission release height, exhaust temperature and velocity, and the time and frequency of emissions. It considers how the pathway of the emissions is affected by wind speed and direction and by buildings, as well as the characteristics of the receptors, including their height and distance from emission sources.

To account for buildings and specific local conditions such as narrow streets enclosed by tall structures (street canyons), the advanced ADMS-Roads Street Canyon Module was used. This module captures a wide range of building geometries, including tall or asymmetric canyons, that can influence concentrations both within and outside the canyon.

Road emissions are well understood and modelled in ADMS. However, ADMS is not specifically designed to represent emissions from moving trains. Therefore, two alternative representations of rail emissions were tested: treating trains as road sources and as line sources, each capturing different aspects of rail emission dispersion.

ADMS is primarily designed to assess long-term pollutant concentrations at a distance from a source, rather than short-term transient effects (less than 1 hour). For most simulations carried out in this study, trains were represented as road sources with intermittent emissions, as is best current practice, since road sources include the effect of turbulence induced by the train moving, which increases the mixing of pollutants at short distances.

However, road sources do not represent plume rise from exhaust temperature and initial velocity, which are important factors for the dispersion of train emissions. Therefore, as a sensitivity test, rail freight emissions

[22] https://www.cerc.co.uk/environmental-software/ADMS-Roads-model.html

were represented as a line source following the alignment of the track, allowing the effects of exhaust temperature and exit velocity to be considered.

Model inputs

London Atmospheric Emissions Inventory (LAEI) 2022

The 2022 LAEI was released in November 2025. It includes comprehensive and accurate estimates of emissions from different sectors, such as road traffic, domestic combustion, industrial processes, and rail traffic for the year 2022. Emissions estimates for road traffic are based on transport modelling and automated traffic counts by Transport for London across the major roads network and on estimates of vehicle km driven by the Department for Transport. Emissions estimates for rail traffic are based on rail activity data from the Network Rail Open Data platform and g/km emission factors from the NAEI, which have been updated based on work previously carried out for RSSB.[23]

The LAEI outputs are considered to be an accurate and robust representation of the baseline emissions for the area of Greater London, which was the primary focus of this project.

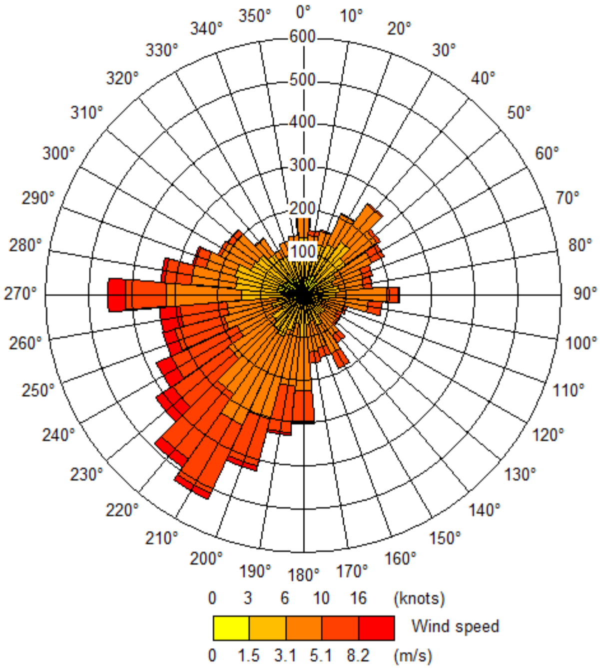
The LAEI 2022 dataset includes traffic activity data (flow and speeds) for major roads in London. These are derived from a strategic transport model (TfL's London Highway Assignment Model, LoHAM) and calibrated using automated traffic counts. These are used to estimate the pollutant emissions from traffic as input to the LAEI 2022 modelled concentrations. The road speeds from this dataset were used as HGV vehicle speeds for the road freight simulations described below.

Meteorology

Meteorological data from Heathrow Airport in 2022 has been used (Figure 14), consistent with the 2022 LAEI modelled concentrations. Note that while the prevailing wind direction is from southwest, dispersion can take place in other directions, as the wind direction is not uniform during the year. Surface roughness length and minimum Monin–Obukhov length parameters typical of outer London were used to represent the characteristics of the urban environment. The surface roughness length describes the influence of buildings, trees, and other surface features on wind flow and turbulence, while the minimum Monin–Obukhov length represents the degree of atmospheric mixing caused by heat released from urban surfaces.

[23] https://www.rssb.co.uk/research-catalogue/CatalogueItem/T1187

Figure 14 Wind speed and direction at Heathrow Airport during 2022



Vehicle emission factors

Rail – Class 66: Emission factors and exhaust parameters by notch for a Class 66 UIC1 locomotive are shown in Table 5 below.

Table 5 Exhaust parameters for Class 66 UIC1 locomotive

Engine traction mode	Temperature [°C]	Volume flux [m^3/s]	NOx [g/s]	PM_{10} [g/s]	f-NO2
Idle	101	0.6422	0.245	0.0288	29.1%
Notch 1	113	0.7415	0.543	0.0084	22.8%
Notch 2	150	0.9021	1.137	0.0347	13.7%
Notch 3	203	1.1596	1.932	0.063	9.3%
Notch 4	247	1.4269	2.638	0.0684	1.8%
Notch 5	285	1.709	3.485	0.0898	2.7%
Notch 6	314	1.7846	4.138	0.1087	2.6%
Notch 7	359	2.4535	6.066	0.1634	3.4%
Notch 8	357	3.0112	7.038	0.1936	2.1%

Rail – Class 70: Emission factors and exhaust parameters by notch for the Class 70 locomotive (which meets the Euro IIIA emissions standard) are shown in Table 6 below.

Table 6 Exhaust parameters for Class 70 locomotive

Engine traction mode	Temperature [°C]	Volume flux [m^3/s]	NOx [g/s]	PM_{10} [g/s]	f-NO_2
Idle	94	0.9161	0.0086	0.0001	42.5%
Notch 1	147	1.3002	0.3454	0.0030	16.2%
Notch 2	198	1.9866	0.7771	0.0068	10.5%
Notch 3	231	2.8244	1.4679	0.0128	7.5%
Notch 4	305	3.0012	2.1587	0.0188	5.2%
Notch 5	354	3.0787	2.8495	0.0248	3.9%
Notch 6	369	3.1596	3.5402	0.0308	3.0%
Notch 7	372	3.2314	4.1447	0.0360	2.4%
Notch 8	382	3.3020	4.7491	0.0413	2.0%

Rail abrasive emissions: SEEsim only calculates particulate emissions resulting from engine combustion. It does not calculate abrasive particulates relating to factors such as braking and wheel and rail wear. In order to ensure comparability with the road alternatives, the approach for the earlier Rail Partners study was followed and included a constant abrasive particulates factor of 0.013 g/vehicle-km for total rail, wheel, and brake wear, which was derived in the RSSB T1187 emission factors project.[24] Based on the literature review conducted for T1187, abrasive particulate emissions for rail are significantly smaller than they are for road (55% to 85% lower), so this approach is thought to be a sufficient approximation.

Road: For the plots in Section 4, a single speed of 53 kph, typical of the London North Circular Road, was used. For the exposure modelling, vehicle speeds from the LAEI 2022 dataset were used for each road link modelled. A Euro class HGV fleet composition typical of outer London in 2022 was assumed for consistency with the LAEI 2022 concentrations. This composition has a very high proportion of Euro VI HGVs, reflecting compliance with the requirements of the London Low Emissions Zone. The emission factors for a single vehicle used for the results in Section 4 are included in Table 7 below, and they were multiplied by the number of vehicles carrying the same load as each freight train**.** These factors include non-exhaust particulate emissions such as tyre wear, brake wear, and road abrasion.

[24] https://www.rssb.co.uk/research-catalogue/CatalogueItem/T1187

Table 7 Emission factors for road freight vehicles. Source: Emission Factor Toolkit (EFT) v.13.1. Fleet composition typical of Outer London, year 2022

Pollutant	Articulated HGV emissions (g/km/s)	Rigid HGV emissions (g/km/s)
NOx	8.84E-05	7.25E-05
PM_{10}	3.81E-05	2.92E-05
$PM_{2.5}$	2.20E-05	1.55E-05

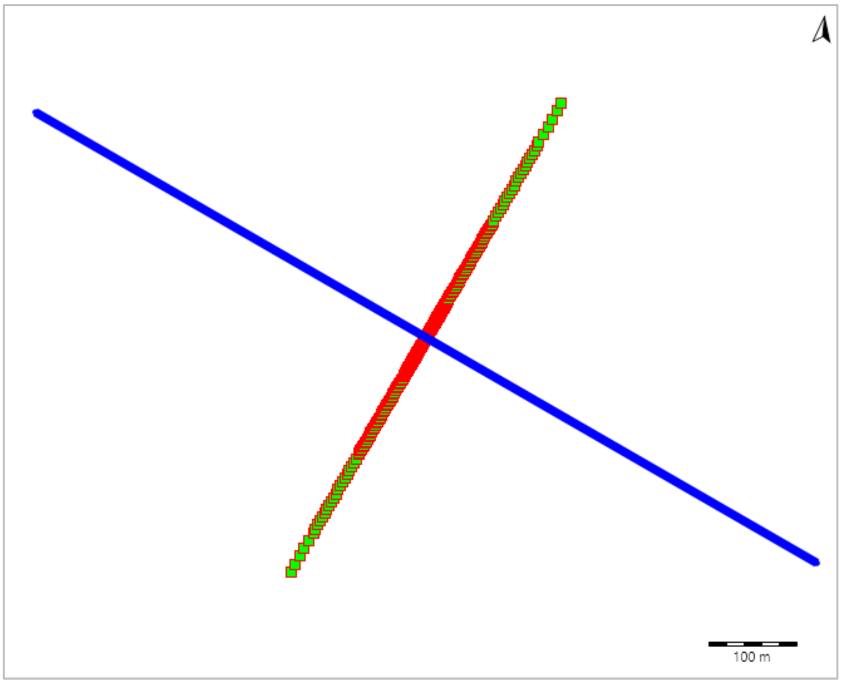
Model set-up

Model geometry

Each simulation was run with only one source (representing the rail or road traffic) as a straight line perpendicular to the prevailing wind direction in order to capture the worst-case effects from wind dispersion (Figure 15).

Model receptor points were placed in a vertical cross-section from the middle of the source directly down-wind at set distances (between 0 m and 200 m from the source centre) and heights (from 0 m to 50 m from ground level), as shown in Figure 15.

Figure 15 Model geometry. Source in blue, receptor points in green



Time-varying emissions

The study considered the impact of a small number of vehicles (a two-way trip by a single freight train per day or an equivalent number of HGVs carrying the same overall load).

To represent the intermittent nature of train operations (a single train passes by receptors only twice per day for a fraction of an hour), the time-varying profile module was used to turn emissions on only for 2 hours each day (6 a.m. and 3 p.m.) and set to zero for the remaining 22 hours. These times of day are realistic for the freight train flows assessed, as they are based on the freight train schedules for the services discussed in Appendix A. Nonetheless, a sensitivity test was carried out with outputs for each hour of the year (emissions on for 24 hours) to estimate the impact on dispersion of changes in meteorological parameters (boundary layer height, wind speed, air temperature) and verify that the times of day chosen do not introduce significant biases in the dispersion, i.e., to ensure that other times of the day would not result in higher concentrations due to meteorological parameters, ensuring that significant worst-case conditions would not be missed.

For HGVs, it was assumed that the traffic flow would follow the same diurnal and weekly pattern as general traffic. A time-varying profile was derived from the Department for Transport's TRA0307: Traffic distribution by time of day and day of the week on all roads in Great Britain for the year 2022.[25]

Pollutants

Annual average concentrations of NOx and PM_{10} were modelled. Concentrations of NOx were then converted to NO_2 as specified below. Since very little information is available on specific $PM_{2.5}$ emission factors by locomotive type or notch (the RSSB T1187 emission factors report[26] recommended use of a flat $PM_{2.5}$:PM_{10} ratio of 74.5% based on available studies), $PM_{2.5}$ was not modelled for this study.

Emissions

For the main simulations, a single set of typical vehicle speeds and emission rates was assumed. As the model runs on a fixed time step of 1 hour, the modelled emissions (in g/km/s) are based on the time the vehicle spends on the track, which is, in turn, dependent on the vehicle speed.

Rail emissions were assigned at a representative release height above ground level, reflecting the typical exhaust position of diesel locomotives, along with the height of the ballast and track. The SEEsim emissions outputs were converted into suitable emission rates for ADMS-Roads and distributed along each rail section.

Road emissions were represented as road sources corresponding to the selected road corridors within each simulation domain. To capture variability in emissions due to traffic patterns, time-varying profiles from DfT statistics (Table TRA0307) were applied to represent typical diurnal and weekly variations.

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[25] https://www.gov.uk/government/statistical-data-sets/road-traffic-estimates-tra

[26] https://www.rssb.co.uk/research-catalogue/CatalogueItem/T1187

Defra's NOx-to-NO_2 calculator v9.1[27] was used to estimate the effect of chemical reactivity in the atmosphere to convert nitrogen oxide (NO) into nitrogen dioxide (NO_2). This was applied to the NOx concentrations modelled in ADMS for both rail and road source simulations.

Typical values for North London in 2022 for background concentrations of NO_2 and regional concentrations above the surface layer for ozone (O_3), NOx, and NO were used.

Values for the primary fraction of NO_2 (f-NO_2, the amount of NO_2 emitted directly by the source) were specified for both freight trains and freight HGVs to feed into the Defra NOx-to-NO_2 tool, as shown in Table 8 below.

Table 8 Fraction of NOx Emitted by Vehicles as NO_2 (by volume)

Source		f-NO_2
Rail (Class 66 UIC1 locomotive)	Idle	0.291
	Notch 1	0.228
	Notch 2	0.137
	Notch 3	0.093
	Notch 4	0.018
	Notch 5	0.027
	Notch 6	0.026
	Notch 7	0.034
	Notch 8	0.021
Road	Rigid HGVs	0.120
	Articulated HGVs	0.124

[27] Defra. (2022). Local Air Quality Management Technical Guidance (England) 2022 (TG22). HMSO. https://laqm.defra.gov.uk/air-quality/featured/uk-regions-exc-london-technical-guidance/

The rail *f*-NO_2 factors have been derived from engine exhaust emissions testing data for a Class 66 UIC1 locomotive. The HGV *f*-NO_2 factors have been sourced from the NAEI as typical for the HGV fleet in London in 2022.[28]

Plume rise (line source) simulations

Plume rise refers to the effect of exhaust temperature (T) and velocity (v) on the shape of the release plume. The height of the plume in turn affects how pollutants disperse in both vertical and horizontal directions. These are important factors to explore in the dispersion of train emissions, as T and v depend on locomotive notch and speed.

ADMS-Roads can simulate these effects for industrial sources, including line sources, but not for road sources. Moreover, in the model, dispersion from line sources is not affected by buildings. For these reasons, a specific set of plume rise simulations was run to understand the sensitivity to this issue. The results of these simulations are presented separately from the main simulations.

Model verification

Since the simulations were based on idealised representations of various typical situations along rail and road routes within London, model verification on these simulations was not carried out, with the understanding that any model bias would affect the results of road and rail sources to the same extent.

Simulation suite

Table 9 below describes the range of building configurations and densities considered for both road and rail simulations. Additional simulations were run to investigate the impact of treating rail as a line source. These were three simulations of Class 66 UIC1 with different combinations of speed and notch, plus one simulation of a different locomotive (Class 70, which meets the Euro IIIa emissions standard).

[28] UK National Atmospheric Emission Inventory. (2023). PrimaryNO2 factors_NAEIBase 2023_v1.xlsx. Available at https://naei.energysecurity.gov.uk/sites/default/files/2024-05/PrimaryNO2%20factors_NAEIBase%202023_v1.xlsx

Table 9 Simulations and their characteristics

Simulation no.	Name	Source treatment	Terrain/buildings	Traffic level	Traffic speed and notch	Other key characteristics
Main simulation set — Rail freight modelled as road source						
Rail 01	Flat Terrain	Rail as a road source	Flat, no buildings	2 trains/day (1 return trip), 6 a.m. outgoing and 3 p.m. return	25 mph (41 kph), Notch 4	One line source, 1 km long and 7 m wide (as a double rail track)
Rail 04	Terrace Houses	As above	Buildings are 7.5 m tall, 25 m from track centreline	As above	As above	Houses on both sides of the rail line form a shallow canyon No other buildings beyond 25 m Very dense (only 10% space between buildings)
Rail 07	Blocks of Flats	As above	Buildings are 21 m tall, 12 m from track centreline	As above	As above	Downwind side only No other buildings beyond 12 m from the rail line Sparse (50% space between buildings)
Rail 02	Viaduct	As above	Elevated track 7.5 m high, no buildings	As above	As above	Source height (viaduct + train) 12.5 m
Rail 03c	Shallow Cutting	As above	7.5 m deep, complex terrain	As above	As above	47 m wide, represented by 128 x 128 terrain grid – 1,500 m a side

Simulation no.	Name	Source treatment	Terrain/buildings	Traffic level	Traffic speed and notch	Other key characteristics
Rail 03b	Deep Cutting	As above	7.5 m deep, Asymmetrical canyon	As above	As above	Street canyon 14 m wide (3.5 m downwind, 10.5 m upwind)
Rail 05	Houses Near Viaduct	As above	Elevated track 7.5 m high, street canyon 7.5 m tall, 25 m from track centreline	As above	As above	Houses on both sides of the rail line form a shallow canyon No other buildings beyond 25 m Very dense (only 10% space between buildings) Source height (viaduct + train) 12.5 m
Rail 06	Houses Near Cutting	As above	Shallow cutting 7.5 m deep, complex terrain. Terrace houses are 7.5 m tall located at edge of cutting (15 m total above track), 25 m from track centreline	As above	As above	47 m wide 128 x 128 terrain grid – 1,500 m a side Canyon on both sides No other buildings beyond 25 m Very dense (only 10% space between buildings)
Rail 08	Viaduct by Flats	As above	Viaduct 7.5 m tall Source height (viaduct + train): 12.5 m Tall buildings 21 m tall 12 m from track centreline	As above	As above	Downwind side only No other buildings beyond 12 m Sparse distribution (50% space between buildings)

Simulation no.	Name	Source treatment	Terrain/buildings	Traffic level	Traffic speed and notch	Other key characteristics
Rail 09	Cutting by Flats	As above	Shallow cutting 7.5 m deep, complex terrain. Tall buildings 21 m tall, 12 m from track centreline	As above	As above	47 m wide 128 x 128 terrain grid – 1,500 m a side Downwind side only No other buildings beyond 12 m Sparse distribution (50% space between buildings)
Main simulation set — Road freight modelled as road source						
Road 01	Flat Terrain	Road source	Flat, no buildings One road source 1 km long 9 m wide (2.5 lanes of dual carriageway) 0 m height	97 rigid HGVs/day (1 return trip)	33 mph (53 kph), Euro VI HGVs (Outer London fleet mix)	Base traffic 1,800 vehicles/hr (8% HGVs) Hourly profile based on 2022 traffic distribution across Great Britain (DfT TRA0307)
Road 04	Road with terrace houses	As above	Road source 10 m wide Terrace houses 7.5 m tall	As above	As above	Canyon 14.5 m from road centreline downwind, 24.5 m upwind No other buildings beyond 25 m Very dense distribution (only 10% space between buildings)
Road 07	Road with Flats	As above	Road source 10 m wide Tall building 21 m tall	As above	As above	Downwind side only No other buildings beyond 12 m

Simulation no.	Name	Source treatment	Terrain/buildings	Traffic level	Traffic speed and notch	Other key characteristics
			16.5 m from road centreline			Sparse distribution (50% space between buildings)
Road 10	Road with Tall Terraces	As above	Road source 7 m wide Terrace houses 10. 5m tall Canyon 10 m from road centreline on both sides	As above	As above	No other buildings beyond 25 m Very dense (only 10% space between buildings)
Sensitivity test — Rail freight modelled as line source						
Line 01	Rail as a Line Source	Rail as a line source	Flat terrain (no buildings) Source at 5 m height (ballast + track + train)	2 trains/day (1 return trip), 6 a.m. outgoing and 3 p.m. return	25 mph (41 kph), Notch 4	One line source, 1 km long and 0.4 m wide (exhaust diameter)
Line 02	Rail as a Line Source (Low Emissions)	As above	As above	As above	40 mph (64 kph), Notch 0 (idling)	As above
Line 03	Rail as a Line Source (High Emissions)	As above	As above	As above	10 mph (16 kph), Notch 8	As above

Simulation no.	Name	Source treatment	Terrain/buildings	Traffic level	Traffic speed and notch	Other key characteristics
Line 01_70	Class 70 as a Line Source	As above	As above	As above	25 mph (41 kph), Notch 4	One line source, 1 km long and 0.26 m wide (exhaust diameter)
Sensitivity test – Abrasive PM_{10} emissions from Rail freight						
AbrPM 01a	Abrasive PM, no dry deposition	Rail as a road source	Flat terrain (no buildings) Source at 0.5 m height (ballast + track)	2 trains/day (1 return trip), 6 a.m. outgoing and 3 p.m. return	Emissions independent of speed No exhaust emissions	One line source, 1 km long and 7 m wide (as a double rail track) Abrasive PM emissions equal to 0.013 g/vehicle/km. 20 vehicles (1 engine, 19 wagons) No dry deposition – no effect from particle size or density
AbrPM 01c	Abrasive PM, with dry deposition	Rail as a road source	As above	As above	As above	Dry deposition module active. Particle diameter 10^{-5} m, Particle density 7,850 kg/m^3 (as steel)

Model outputs

Model outputs include a combination of transect-based receptor points for each simulation and gridded concentration points for defined simulations. Transects were set perpendicular to the road and rail alignments, with receptor points spaced at defined intervals and multiple heights to capture the variation in concentrations with distance and elevation from the source. Gridded outputs were generated at an appropriate horizontal resolution (i.e., 10 m × 10 m) to visualise pollutant dispersion and concentration contours across selected domains. This dual output structure provides both detailed quantitative data for comparative analysis and spatial maps illustrating concentration gradients and incremental (delta) changes between the road and rail freight scenarios.

The associated output files contain (for each simulation):

- **inputs:** ADMS-Roads modelling outputs (receptors, locations, output concentrations)
- **modelling results:** assumptions that apply to the whole dataset (emission rates, baseline concentrations, primary NO_2 fraction)
- **concentrations:** NOx and PM_{10} concentrations derived by scaling model outputs by emission factors, with NO_2 concentrations converted using Defra NOx-to-NO_2 tool (tall data format, used for contour plots)
- **wide format:** same data as 'concentrations', pivoted to wide data format (distance as columns, height as rows) and colour-coded
- **fall-off curve:** extracts from 'wide format' at set heights for charting as fall-off curves (concentration versus distance).

Assessment of impact magnitude

The impact magnitude criteria in the IAQM guidance are summarised below in Table 10. The criteria are made up of two steps: firstly, to assess the magnitude of change in concentration relative to the air quality objective (AQO), and secondly, to assess the percentage above/below the AQO based upon the total modelled concentration at a given location or receptor. These two values are then combined to obtain the impact descriptor.

Table 10 IAQM impact magnitude criteria

Long-term average concentration at receptor in assessment year	Percentage (%) change in concentration relative to annual mean air quality objective (AQO)			
	<1	2–5	6–10	>10
75% or less of AQO	Negligible	Negligible	Minor	Moderate
76%–94 % of AQO	Negligible	Minor	Moderate	Moderate
95%–102 % of AQO	Minor	Moderate	Moderate	Substantial
103%–109 % of AQO	Moderate	Moderate	Substantial	Substantial
110% or more of AQO	Moderate	Substantial	Substantial	Substantial

Notes:

Where concentrations increase, the impact is described as adverse, and where they decrease, the impact is described as beneficial.

*% change is rounded to nearest whole number. Where the % change is less than 0.5%, the impact will be 'negligible'.

Appendix C: Route exposure modelling supporting information

Population allocation methodology

Sensitive receptors within 200 m of road routes and rail corridors were identified using the Ordnance Survey Open UPRN (Unique Property Reference Number)[29] and Office for National Statistics (ONS) population density[30] datasets. These freely available datasets provide information on the number and spatial distribution of properties along the selected corridors, so they can support subsequent exposure and impact analyses.

The average population in each Census Output Area was allocated to each Ordnance Survey OpenUPRN address point, excluding any properties falling within Network Rail boundaries. Building configurations were taken from OpenStreetMap, and a conservative assumption was made for where the closest frontage of buildings lay with the spatial bands, i.e., 25 m, 50 m, 100 m, 150 m, 200 m of the road or rail line. The population in tall buildings (i.e., multiple addresses at the same location) was assumed to be at half the height of a typical tall building.

Population distributions for each band and for each route are shown in Figure 16 below.

The impact score method is based on the Department for Transport's Environmental Impact Appraisal (TAG):[31]

- For each segment of road or rail, the pollutant concentration contribution at the specified point in the band was multiplied by the population within that band to give population weighted concentrations.
- This was carried out for each of the four bands and the results added together to give a total for each segment.
- The scores for each segment were then added together to provide the overall score for the route.

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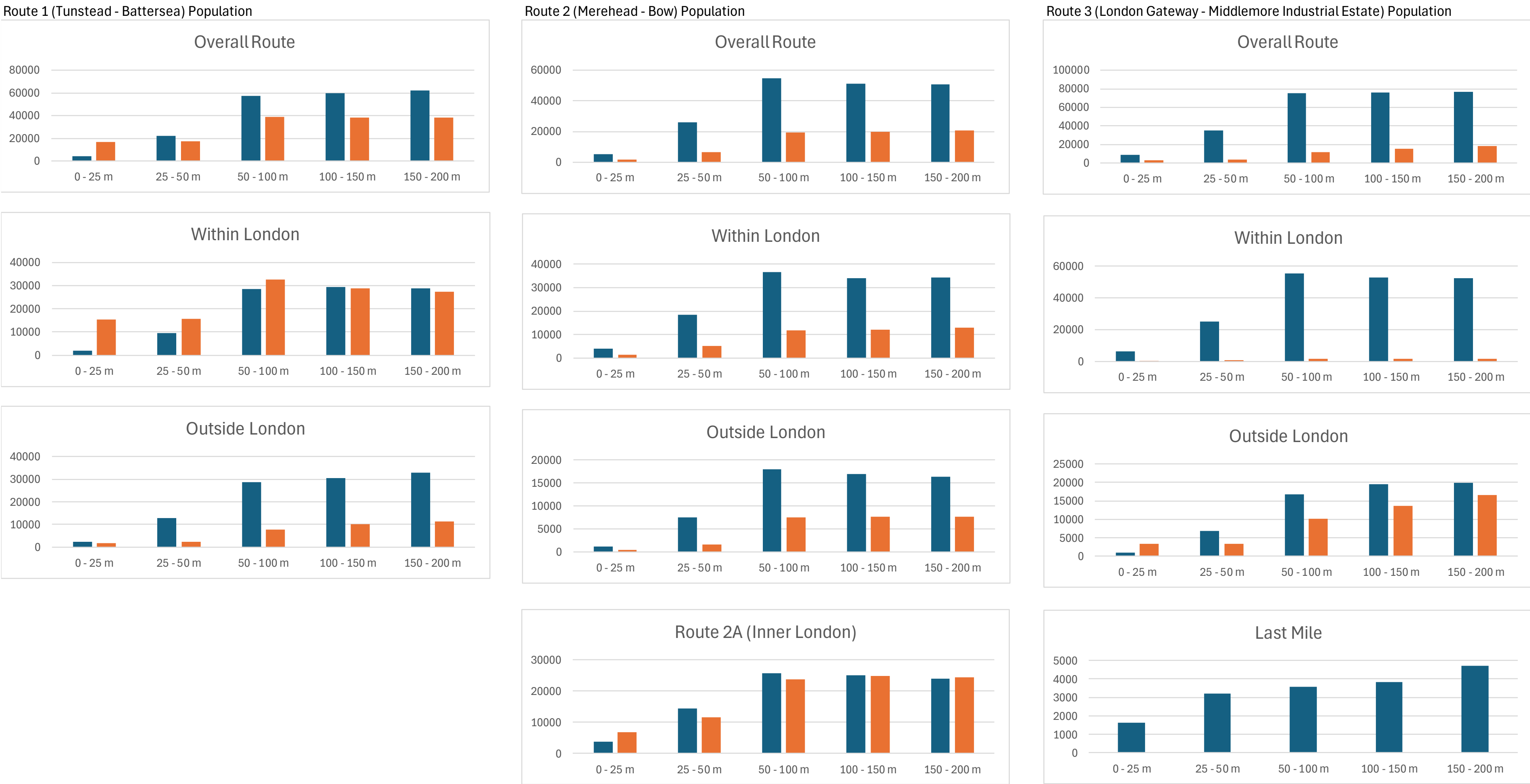
[29] https://www.ordnancesurvey.co.uk/products/os-open-uprn

[30] https://www.ons.gov.uk/datasets/TS006/editions/2021/versions/4

[31] TAG Unit A3 Environmental Impact Appraisal, pages 21 and 23. Available at: https://assets.publishing.service.gov.uk/media/69984e7ba58a315dbe72c082/tag-unit-a3-environmental-impact-appraisal.pdf

Figure 16 Population distributions along the study routes (number of people) (rail = blue, road = orange)



Existing ambient pollutant concentrations

Modelling existing ambient pollutant concentrations, i.e., pollution already present in the air, including contributions from other existing road and rail traffic and other sources, was not the focus of this project. To provide context and support the assessment (e.g., to inform the NOx to NO_2 conversion calculations), information publicly available from regional and national modelling was used to characterise existing pollutant concentrations.

Within Greater London, the 2022 LAEI includes ground level concentrations of key pollutants NOx, NO_2, PM_{10}, and $PM_{2.5}$ for year 2022 at 20-m spatial resolution. These concentrations take into account the regional background and all emissions included in the LAEI, including the existing traffic on the road and rail routes considered in this project.

Outside of London, the air pollution background concentration maps published by Defra and the Devolved Administrations were used. These include contributions from natural emissions and human activities, on a 1-km spatial resolution across the United Kingdom. They are an accurate representation of the regional background and a good indication of the change in concentration at regional scale. The air pollution background concentration maps do not have the same level of spatial resolution or the same level of detail in the sources included, as the LAEI concentrations. Therefore, care should be taken in comparing them directly. They are included in this study as an indication of existing concentrations across the routes area outside London.

As discussed in Section 2.4, existing ambient pollutant concentrations are generally expected to be the highest in densely populated urban areas. This is reflected in the existing ambient pollutant concentrations presented in Figure 17 and Figure 18. Within London, annual mean NO_2 existing ambient concentrations are typically in the range of approximately 20–35 $\mu g/m^3$, whereas concentrations outside London are generally below 20 $\mu g/m^3$. Similarly, annual mean PM_{10} existing ambient concentrations within London are typically around 15–20 $\mu g/m^3$, compared with generally less than 15 $\mu g/m^3$ outside London.

Figure 17 Existing NO_2 concentrations along the study routes (µg/m^3) (rail = blue, road = orange)

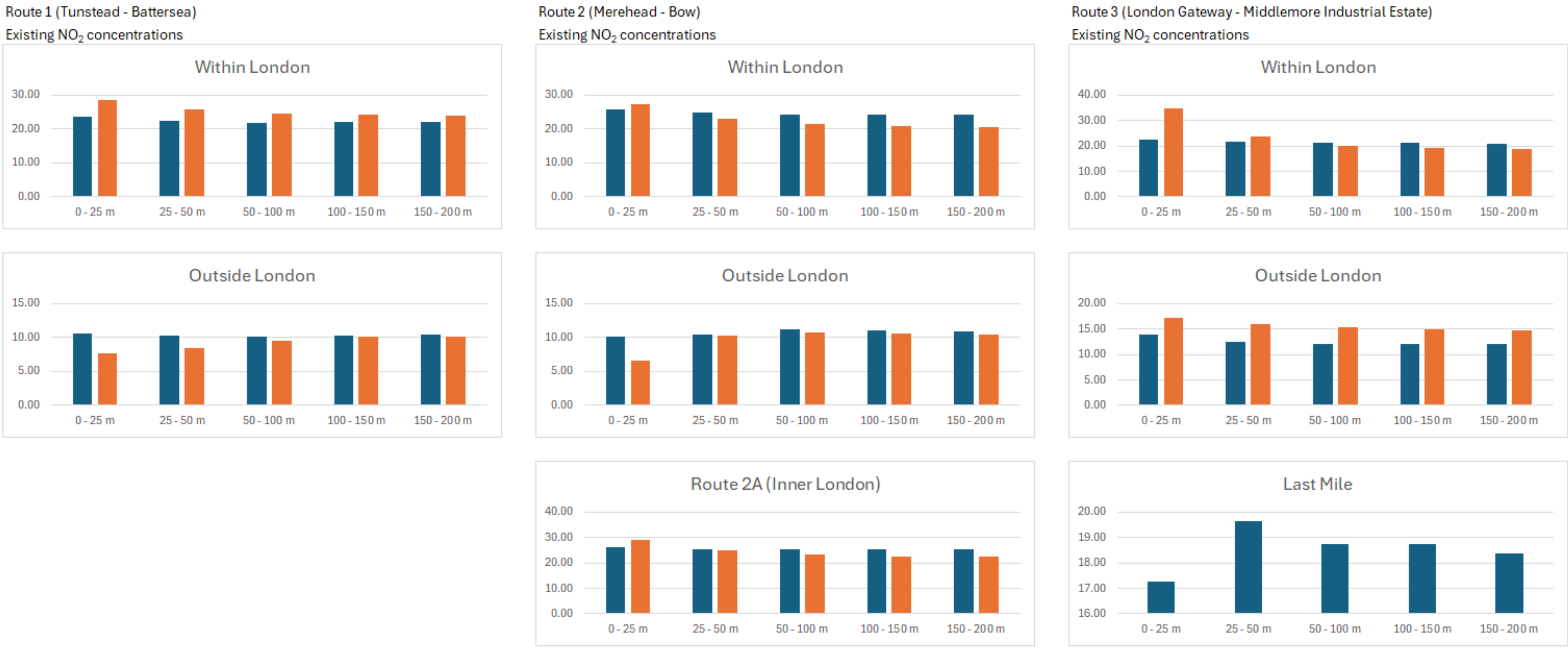


Figure 18 Existing PM_{10} concentrations along the study routes ($\mu g/m^3$) (rail = blue, road = orange)

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