Lower Thames Crossing

Thurrock Council Local Impact Report

Appendix D – Council: Borough Wide Air Quality and Noise Modelling

Lower Thames Crossing

Thurrock Council Local Impact Report

Appendix D – Annex 1 Thurrock Council: Borough Wide Air Quality Modelling

TECHNICAL NOTE

Job Name: Thurrock Council – Technical Support (Air Quality)

Job No: 332510966

Note No: AQ001

Date: December 2022

Prepared By: Daniel Francis/Philip Branchflower

Subject: **Thurrock Council: Borough Wide Air Quality Modelling**

1. Introduction

1.1. Stantec UK Ltd. has been commissioned by Thurrock Council to generate visualisations of the potential air quality impacts associated with the Lower Thames Crossing (LTC) development.

1.2. This note has been prepared by Stantec to provide a summary of the basis of the images illustrating predicted annual average NO_2 and $PM_{2.5}$ concentrations across Thurrock (with and without the Lower Thames Crossing (LTC) scheme).

1.3. These images combine appropriate DEFRA predicted background concentrations with modelled contributions from road traffic which have been generated through combining traffic modelling data, DEFRA published pollutant emission factors for road traffic and atmospheric dispersion modelling (undertaken in accordance with DEFRA TG(16) guidance).

1.4. The following images are provided with this technical note:

- Figure 1: NO_2 DM. Predicted annual average NO_2 concentration: Do-minimum scenario; opening year traffic without LTC.
- Figure 2: NO_2 DS. Predicted annual average NO_2 concentration: Do-something scenario; opening year traffic with LTC.
- Figure 3: NO_2 'change'. Predicted change in annual average NO_2 concentration between DM and DS scenarios i.e. as a consequence of LTC.
- Figure 4: $PM_{2.5}$ DM. Predicted annual average $PM_{2.5}$ concentration: Do-minimum scenario; opening year traffic without LTC.
- Figure 5: $PM_{2.5}$ DS. Predicted annual average $PM_{2.5}$ concentration: Do-something scenario; opening year traffic with LTC.
- Figure 6: $PM_{2.5}$ 'change'. Predicted change in annual average $PM_{2.5}$ concentration between DM and DS scenarios i.e. as a consequence of LTC.

1.5. The methodology applied in this assessment is considered to comply with the requirements of relevant guidance (DEFRA, IAQM) for assessing air quality impacts of schemes.

1.6. It is important to note that National Highways (NH) will apply a different approach (based on their DMRB guidance) to both model inputs and interpretation of the results.

1.7. Further technical detail regarding the basis of these predictions, and to enable interpretation and understanding of them, is provided in the following sections.

2. Relevant Assessment Levels

2.1. The Air Quality (England) Regulations 2000 (AQR) defined National Air Quality Objectives (NAQOs, a combination of concentration-based thresholds, averaging periods and compliance dates) for a limited range of pollutants. Subsequent amendments were made to the AQR in 2001 and 2002 to incorporate 'limit values' and 'target values' for a wider range of pollutants as defined in European Union (EU) Directives.

2.2. These amendments were consolidated by the Air Quality Standards Regulations 2010 (AQSR) (with subsequent amendments most notably in 2016 and for the devolved administrations), which transposed the EU's Directive on ambient air quality and cleaner air for Europe (2008/50/EC).

2.3. Following the Transition Period after the UK's departure from the EU in January 2020, the Air Quality (Amendment of Domestic Regulations) (EU Exit) Regulations 2019 (and subsequent amendments for the devolved administrations) have amended the AQ Standards Regulations 2010 to reflect the fact that the UK has left the EU. The Environment (Miscellaneous Amendments) (EU Exit) Regulations 2020 amended the $PM_{2.5}$ limit value in the AQSR to 20 µg/m^3 (from 25 µg/m^3).

2.4. The AQOs and limit values, which represent the legal requirements in relation to ambient air quality, are shown in **Table2.1**.

Table2.1: Relevant Air Quality Objectives / Limit Values

Pollutant	Time Period	Objectives	Source
NO_2	1-hour mean	200 µg/m^3 not to be exceeded more than 18 times a year	AQO and AQSR limit value
	Annual mean	40 µg/m^3	AQO and AQSR limit value
$PM_{2.5}$	Annual mean	20 µg/m^3	AQSR limit value

2.5. The 2019 Clean Air Strategy includes a commitment to set a "*new, ambitious, long-term target to reduce people's exposure to $PM_{2.5}$*" which the Environment Act has committed the Secretary of State to setting. This limit is the subject of ongoing consultation and DEFRA has proposed an annual mean concentration target of 10µg/m^3 by 2040. Additionally, the Mayor of London has committed to meeting the World Health organisation (WHO) guideline of 10 µg/m^3 by 2030 within London.

2.6. The WHO guidelines (presented in **Table2.2.2**) are not currently in the form of regulations and there is no legal requirement to meet them, however they are based on the latest evidence of the health effects of air pollution.

Table2.2: WHO Guidelines

Pollutant	Time Period	Interim Target (µg/m^3)				AQG Level (µg/m^3)
		1	2	3	4	
Nitrogen Dioxide (NO_2)	1-hour Mean	-	-	-	-	200
	24-hour Mean	120	50	-	-	25
	Annual Mean	40	30	20	-	10
Fine Particles ($PM_{2.5}$)	24-hour Mean	75	50	37.5	25	15
	Annual Mean	35	25	15	10	5

2.7. Therefore, whilst the AQSR objectives and limit values represent the current legal limits, when considering future requirements and potential health effects, consideration should also be given to the forthcoming Environment Act requirements for $PM_{2.5}$ and WHO guidelines.

3. Significance of Impacts

3.1. There is no official guidance in the UK on how to assess the significance of the air quality impacts of a new development on existing receptors. The approach developed by EPUK and the IAQM (EPUK / IAQM, 2017) is generally applied in the UK and considers the change in air quality (due to a development) in combination with baseline concentrations at the receptors. The guidance sets out three stages: determining the magnitude of change at each receptor, describing the impact, and assessing the overall significance. Impact magnitude relates to the change in pollutant concentration; the impact description relates this change to the air quality objective and is shown in **Table 3.1**.

Table 3.1: Impact Significance Criteria

Long term average Concentration at receptor in assessment year	% Changes in Concentration with development in relation to NAQO / Limit Value			
	1* (negligible)	**2-5 (small)**	**6-10 (medium)**	**>10 (large)**
> 110 % [a]	Moderate	Substantial	Substantial	Substantial
>102% - ≤110% [b]	Moderate	Moderate	Substantial	Substantial
>95% - ≤102% [c]	Slight	Moderate	Moderate	Substantial
>75% - ≤95% [d]	Negligible	Slight	Moderate	Moderate
≤75% [e]	Negligible	Negligible	Slight	Moderate

Where concentrations increase the impact is described as adverse, and where it decreases as beneficial.
% change rounded to nearest whole number. Where the % change is 0 (i.e. Less than 0.5%) the impact will be Negligible.

3.2. The IAQM guidance states that the overall assessment of significance should be based on professional judgement, taking into account factors including:

- The number of properties affected by 'slight', 'moderate' or 'substantial' adverse air quality impacts and a judgement on the overall balance;
- The magnitude of the changes and the descriptions of the impacts at the receptors;
- Whether or not an exceedance of a NAQO or limit value is predicted to arise in the operational study area (where there are significant changes in traffic) where none existed before, or an exceedance area is substantially increased;
- The uncertainty, comprising the extent to which worst-case assumptions have been made; and
- The extent to which a NAQO or limit value is exceeded.

3.3. Therefore, where impacts at an individual receptor are classified as 'negligible' or 'slight', effects would typically be considered 'not significant'. However, where 'moderate' or 'substantial' adverse impacts are identified at individual receptors, the overall effect needs to be considered in the round taking into account the changes at all of the modelled receptor locations, with a judgement made as to whether the overall air quality effect of the development is 'significant' or not.

4. Air Quality Modelling Methodology

Traffic Data

4.1. The traffic data (including road geometry) utilised was extracted from a cordon model of the Lower Thames Area Model (LTAM) covering the Borough of Thurrock, known as the Thurrock Area Cordon Model (TACM) provided by NH in July 2021. Whilst NH have recently updated the LTAM and TACM, this data was not available at the time of the air quality modelling study/

4.2. The base (2016) model and the opening year with and without LTC (2029) have been utilised to calculate 24-hour Annual Average Daily Traffic flows per link, speed and % HDV within Thurrock as required for the AQ modelling.

4.3. This has utilised data for the AM, IP and PM peak periods using what is known as Actual Flows in the Saturn model, this takes into account delay at junctions to provide a better understanding of traffic impacts on the highway. 24-hour AADT conversion factors have been extracted from the 'Lower Thames Crossing – Traffic Forecasting Report' from the 2018 Statutory Consultation.

Vehicle Emissions

4.4. Emissions have been calculated using the Emission Factor Toolkit (EFT) v11 (DEFRA, 2020c), which utilises NOx emission factors taken from the European Environment Agency (EEA) COPERT 5.3 emission tool. Road vehicular emissions are primarily associated with the exhaust emissions but also include particles generated from abrasion (of tyres, brakes and road). The EFT allows users to calculate road vehicle pollutant emission rates for NOx, PM_{10} and $PM_{2.5}$ (exhaust and brake, tyre, and road wear) for a specified year, road type, vehicle speed and vehicle fleet composition.

4.5. For the base year, the EFTv11 was modified for a 2016 base fleet (from previous EFT versions) and for the 'opening year' scenario, 2026 emission factors have been applied. This is considered to provide an appropriately conservative assessment considering the uncertainties regarding future vehicle emission factors.

Dispersion Modelling

4.6. Emissions from road vehicles and their resultant concentrations have been predicted with default road widths at a 1.5m elevation across Thurrock using the Rapidair Model utilising a 3m resolution based on meteorological data from Gravesend-Broadness, City and Southend monitoring stations and upper air meteorology from the Herstomonceux monitoring station, as the closest available datasets.

Model Verification

4.7. Most NO_2 is produced in the atmosphere by the reaction of nitric oxide (NO) with ozone. It is therefore most appropriate to verify the model in terms of primary pollutant emission of nitrogen oxides (NOx = NO + NO_2). The model has been run to predict the 2016 annual mean road-NOx contribution at 28 'Roadside' monitoring locations within Thurrock.

4.8. A primary adjustment factor of 3.9552 has been determined as the slope of the best fit line between the modelled road NOx contribution and the 'measured' road-NO_x (which is calculated from the measured and background NO_2 concentrations within DEFRA's NO_x from NO_2 calculator (DEFRA, 2019e)), forced through zero. This factor has then been applied to the raw modelled road-NO_x concentration to provide adjusted modelled road-NO_x concentrations.

4.9. The total NO_2 concentrations have then been determined by combining the adjusted modelled road-NO_x concentrations with the background NO_2 concentration within DEFRA's NO_x from NO_2 calculator (DEFRA, 2019e). A secondary adjustment factor of **0.9984** has then been calculated as the slope of the best fit line applied to the adjusted data and forced through zero.

4.10. The calculated Root Mean Square Error (RMSE) for this verification (6.6μg/m^3) lies within the range considered to be acceptable by DEFRA (DEFRA, 2018a) (4 – 10).

4.11. Due to the limited $PM_{2.5}$ monitoring data available, the primary adjustment factor calculated of NO_2 has been applied to the modelled road-$PM_{2.5}$ concentrations.

Post Processing

4.12. The verified road-NO_x has been converted to NO_2 using Defra's NO_x to NO_2 Calculator (v6.0 for baseline verification and v8.0 for future year scenarios) and a random point regression method. 5,681 random points within Thurrock were sampled for both interpolated NO_2 background (with motorway, primary and trunk contribution removed) and the interpolated adjusted NOx value. This is then converted to road NO_2 values via a regression of these points which is then applied across the model domain.

4.13. The verified NO_2 and $PM_{2.5}$ concentrations have then been added to interpolated (via a kriging methodology with a 12-point spherical semi-variogram) DEFRA background data extracted from Defra's 2018-based background maps for 2026.

5. Summary of Modelling Results

5.1. The currently legal limit for NO_2 is 40μg/m^3 (annual average) which the modelling predicts (Figure 1 & Figure 2) is only exceeded within close proximity to the M25/Dartford Crossing and A13 (and LTC) as well as some hotspots within Thurrock which are the focus of existing Air Quality Management Areas declared by Thurrock Council.

5.2. A similar trend is apparent for $PM_{2.5}$ (Figure 4 & Figure 5) with the current legal limit of 20 μg/m^3 (annual average) only exceeded in close proximity to the M25/Dartford Crossing and A13 (and LTC), however DEFRA are proposing to reduce this limit to 10 μg/m^3 and reduce public exposure by 35% (by 2040) and the World Health Organisation (WHO) have set an Air Quality Guideline of 5 μg/m^3 for the protection of public health.

5.3. The changes in pollutant concentrations (between the DM and DS situations) are presented in Figure 3 (for NO_2) and Figure 6 (for $PM_{2.5}$) respectively. The magnitude of these changes has been classified (based on the current legal limits) as shown in **Table 5.1**:

Table 5.1: Magnitude of Changes Impact

Magnitude of Impact in Air Quality Terms	Predicted change in annual average NO_2 concentration (μg/m^3)	Predicted change in annual average $PM_{2.5}$ concentration (μg/m^3)
Large change (increase or decrease)	4	2
Medium change (increase or decrease)	2	1
Small change (increase or decrease)	0.4	0.2
Negligible change	<0.4	<0.2

5.4. It is apparent from the modelling that there will be reduction in pollutant concentrations in proximity to the M25/Dartford Crossing and sections of the A13 where traffic flows decrease as a result of the LTC, although the changes are classified as 'small' beyond 100m from the carriageway.

5.5. However, the predicted increase in pollutant concentrations in proximity to the LTC itself, and the junction with the A13 are greater (both numerically and extent) with ‘medium’ changes extending across a band which is 500-1,000m wide for NO_2 (Figure 3) and 250-500m wide for $PM_{2.5}$ (Figure 6).

5.6. In relation to $PM_{2.5}$, this ‘medium change’ band equates to a 1 µg/m^3 increase, the implications of which should also be considered in the context of forthcoming UK targets of 10 µg/m^3 and to reduce public exposure to $PM_{2.5}$ by 35% by 2040.

5.7. From analysis of the modelled impacts, the numbers of residential properties (from OS Addressbase data) experiencing a change (increase or decrease) in modelled NO_2 and $PM_{2.5}$ concentrations are summarised in the following Table:

Table 5.2: Property Count with Predicted Magnitude of Change in Annual Average NO_2 and $PM_{2.5}$ concentration

Predicted Change	Number of Residential Properties with Predicted Increased Concentration	Number of Residential Properties with Predicted Decreased Concentration
‘Small’ change in NO_2	18,052	9,343
‘Medium’ change in NO_2	1,863	42
‘Large’ change in NO_2	124	1
‘Small’ change in $PM_{2.5}$	8,782	3,474
‘Medium’ change in $PM_{2.5}$	117	2
‘Large’ change in $PM_{2.5}$	48	0

5.8. These predicted changes in annual average NO_2 and $PM_{2.5}$ concentrations at residential properties have been considered alongside Index of Multiple Deprivation (IMD 2019 by LSOA) as summarised in the following tables.

Table 5.3: Property Count by IMD quintile with Predicted Change in Annual Average NO_2 concentration

IMD quintile	Most Deprived				Least Deprived	Total
	0-20%	**20-40%**	**40-60%**	**60-80%**	**80-100%**	
No of Properties with Predicted Reduction in NO_2 Concentrations	171	2,739	1,693	1,228	3,555	9,386
No of Properties with no appreciable change in predicted NO_2 concentrations	4248	15,799	8,032	8,671	2,844	39,594
No of Properties with Predicted Increase in NO_2 Concentrations	719	7,140	3,630	5,408	800	17,697

Table 5.4: Property Count by IMD quintile with Predicted Change in Annual Average PM2.5 concentration

IMD quintile	Most Deprived				Least Deprived	Total
	0-20%	**20-40%**	**40-60%**	**60-80%**	**80-100%**	
No of Properties with Predicted Reduction in $PM_{2.5}$ Concentrations	283	836	1,262	119	976	3,476
No of Properties with no appreciable change in predicted $PM_{2.5}$ concentrations	6,028	20,861	10,200	13,284	6,223	56,596
No of Properties with Predicted Increase in $PM_{2.5}$ Concentrations	1,169	3,981	1,893	1,904	0	8,947

6. Discussion of Predicted Impacts

6.1. It is apparent that existing residential properties in areas such as Orsett, Orsett Heath, Chadwell St. Mary and East Tilbury (as well as numerous individual dwellings in proximity to the LTC) will experience an increased air pollution burden as consequence of LTC, with properties in proximity to sections of the A13 and M25 experiencing a decrease.

6.2. The analysis of residential properties indicates that the number predicted to experience an increase (‘small’, ‘medium’ and ‘large’) in annual average NO_2 and $PM_{2.5}$ concentrations is substantially greater than the number of properties predicted to experience decreases.

6.3. Furthermore, the analysis of the predicted changes in annual average NO_2 and $PM_{2.5}$ concentrations at residential properties alongside Indices of Multiple Deprivation (IMD) indicate that the air pollution impacts of LTC are not equally distributed; residential properties within deprived areas of Thurrock are more likely to experience increased concentrations of NO_2 and $PM_{2.5}$ whereas residential properties within the least deprived quintile are more likely to experience decreases.

6.4. Whilst the predicted annual average NO_2 impacts are unlikely to exceed the NAQO (of 40 $\mu g/m^3$), the predicted levels throughout Thurrock are in excess of the WHO AQG (of 10 $\mu g/m^3$), this is largely due to background concentrations. In the vicinity of the LTC, these predicted impacts will result in non-compliance with the WHO Interim Target 3 of 20$\mu g/m^3$ at residential properties in proximity to the LTC (as shown in Figure 2).

6.5. The predicted $PM_{2.5}$ impacts do not result in exceedances of the current limit value (of 20 $\mu g/m^3$). However, large areas of Thurrock are predicted to exceed the forthcoming 10$\mu g/m^3$ target (to be achieved by 2040) and therefore any impact from LTC will further exacerbate $PM_{2.5}$ pollution.

6.6. Whilst $PM_{2.5}$ is significantly influenced by background concentrations, where the predicted $PM_{2.5}$ impacts due to LTC alone exceed 1$\mu g/m^3$ (10% of the 2040 target or 20% of the WHO guidelines) as shown by the ‘medium increase’ band in Figure 6; this would be of particular concern for potential impacts on health as $PM_{2.5}$ is a non-threshold pollutant and no ‘safe’ level of exposure has been defined.

6.7. These predictions are based on ‘opening year’ traffic flows and it should be recognised that traffic is predicted to increase significantly within the first 15 years. The rate of renewal of vehicles and uptake of Electric Vehicles (EV) will contribute to anticipated reductions in NOx emissions from road transport; however this is unlikely to result in any noticeable decrease in $PM_{2.5}$ emissions (and heavier weights of EV could result in increased emissions). There is uncertainty as to the rate of this change and whether any decrease in NOx emissions will outweigh the growth in traffic flows using the LTC. Therefore, the duration of these predicted impacts is uncertain and $PM_{2.5}$ impacts due to LTC are likely to increase further in future years with increased traffic flows using the LTC.

6.8. These predicted impacts also need to be considered in relation to the potential introduction of new sensitive receptors such as residential properties and schools as part of the Thurrock Local Plan as the introduction of new receptors into areas at risk of significant adverse impacts will need to be avoided.

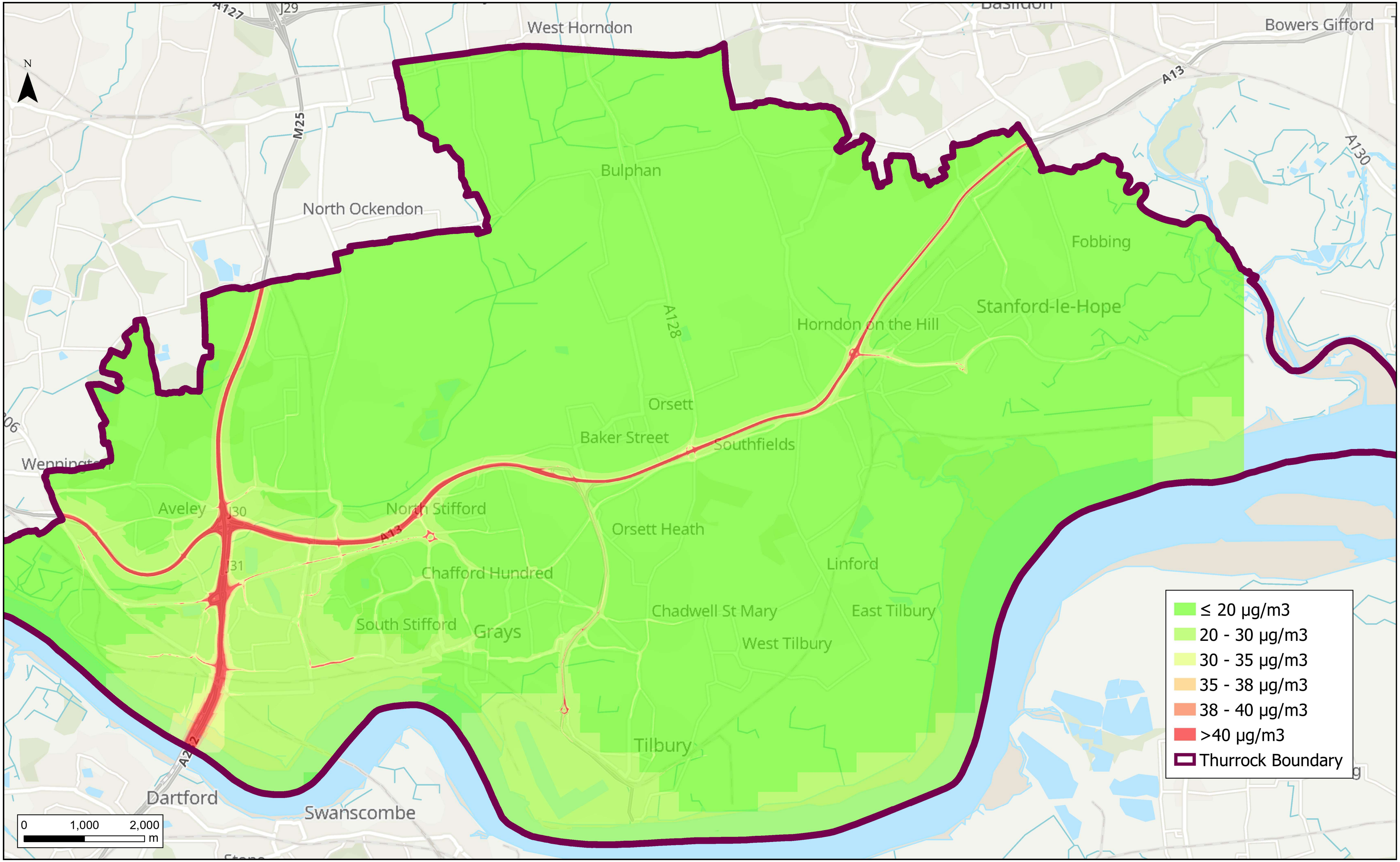
6.9. Given the potential for adverse health impacts due to NO_2 and $PM_{2.5}$ at levels well below the current AQO (or limit values) and the magnitude of the predicted impacts, appropriate mitigation measures should be investigated. This should include consideration of mitigation measures related to the source (i.e., speed limit reduction, encouragement of EV uptake, influencing driver behaviour etc), pathway (i.e. alignment and use of barriers) and receptor (i.e. filtration and awareness raising) as recommended in Highway England Research[1].

6.10. No such evidence has been presented to demonstrate that the efficacy and practicability of options to mitigate the air quality impacts of operational traffic have been considered. NH rely instead on the framework set by their DMRB LA105 guidance which focusses solely on exceedances of the NAQOs.

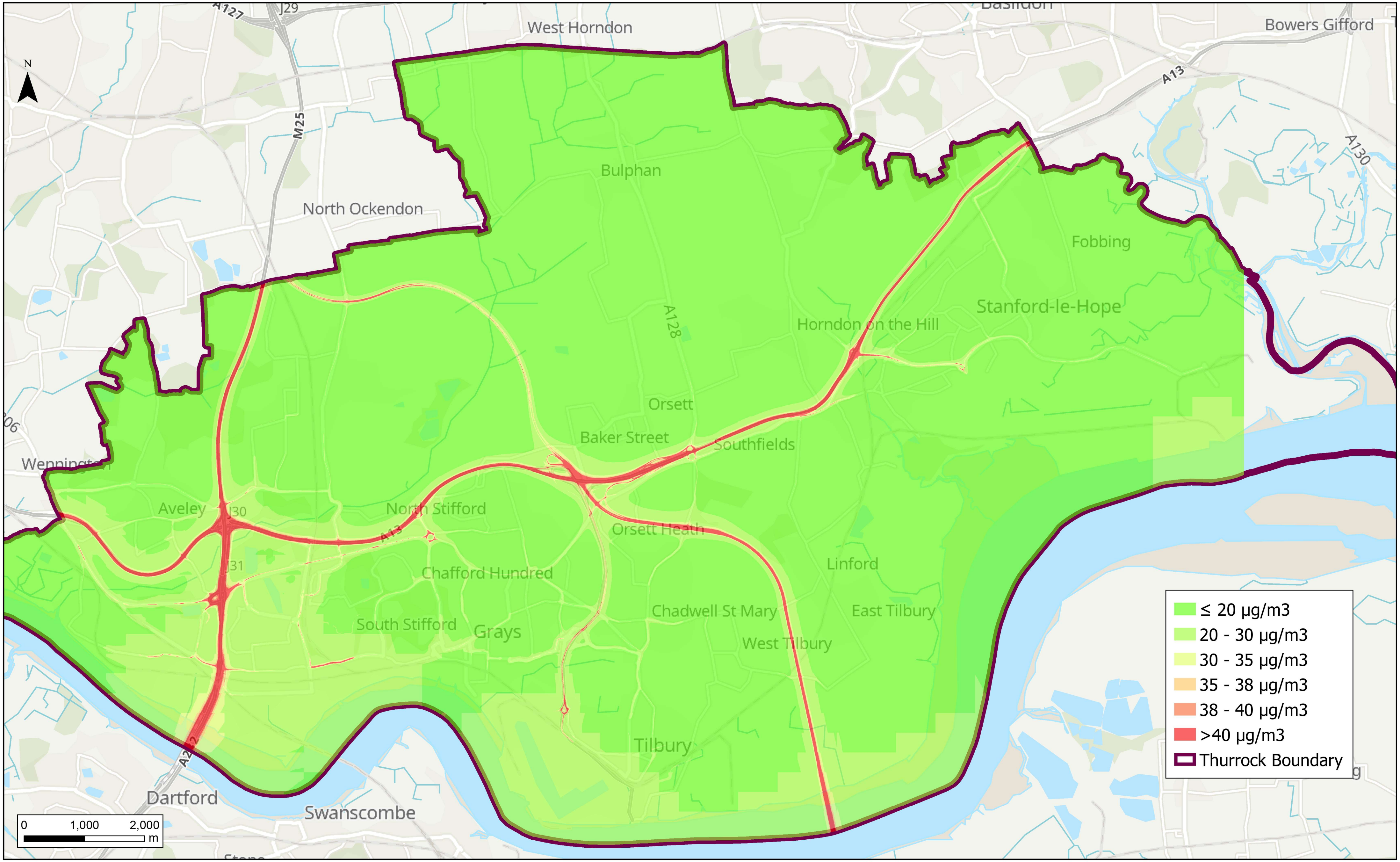
DOCUMENT ISSUE RECORD

Technical Note No	Rev	Date	Prepared	Checked	Reviewed (Discipline Lead)	Approved (Project Director)
Thurrock Council – AQ Modelling	-	July 2022	DF/PB	CB	PB	-
Thurrock Council – AQ Modelling	-	October 2022	DF/PB	CB	PB	KM
Thurrock Council – AQ Modelling		Decemb er 2022	DF/PB	CB	PB	KM

[1] Highways England - Summary of research projects to improve air quality on or close to the strategic road network, December 2019. (available from : https://assets.highwaysengland.co.uk/Corporate+documents/FINAL+-+HE+Research+Projects+to+Improve+Air+Quality.pdf



Stantec	Client THURROCK COUNCIL	**Thurrock Council Air Quality Modelling** Predicted 'Do-minimum' annual average NO_2 concentration.	GB Topographic: Contains OS data © Crown Copyright and database right 2020 Contains data from OS Zoomstack	1:80,000 @ A4 Date: 10/05/2022 Drawn: pbranchflower Checked:PB Figure 01 Rev A



Stantec	Client THURROCK COUNCIL	**Thurrock Council Air Quality Modelling** Predicted 'Do-something' (i.e. with LTC) annual average NO_2 concentration.	GB Topographic: Contains OS data © Crown Copyright and database right 2020 Contains data from OS Zoomstack	1:80,000 @ A4	Date: 10/05/2022
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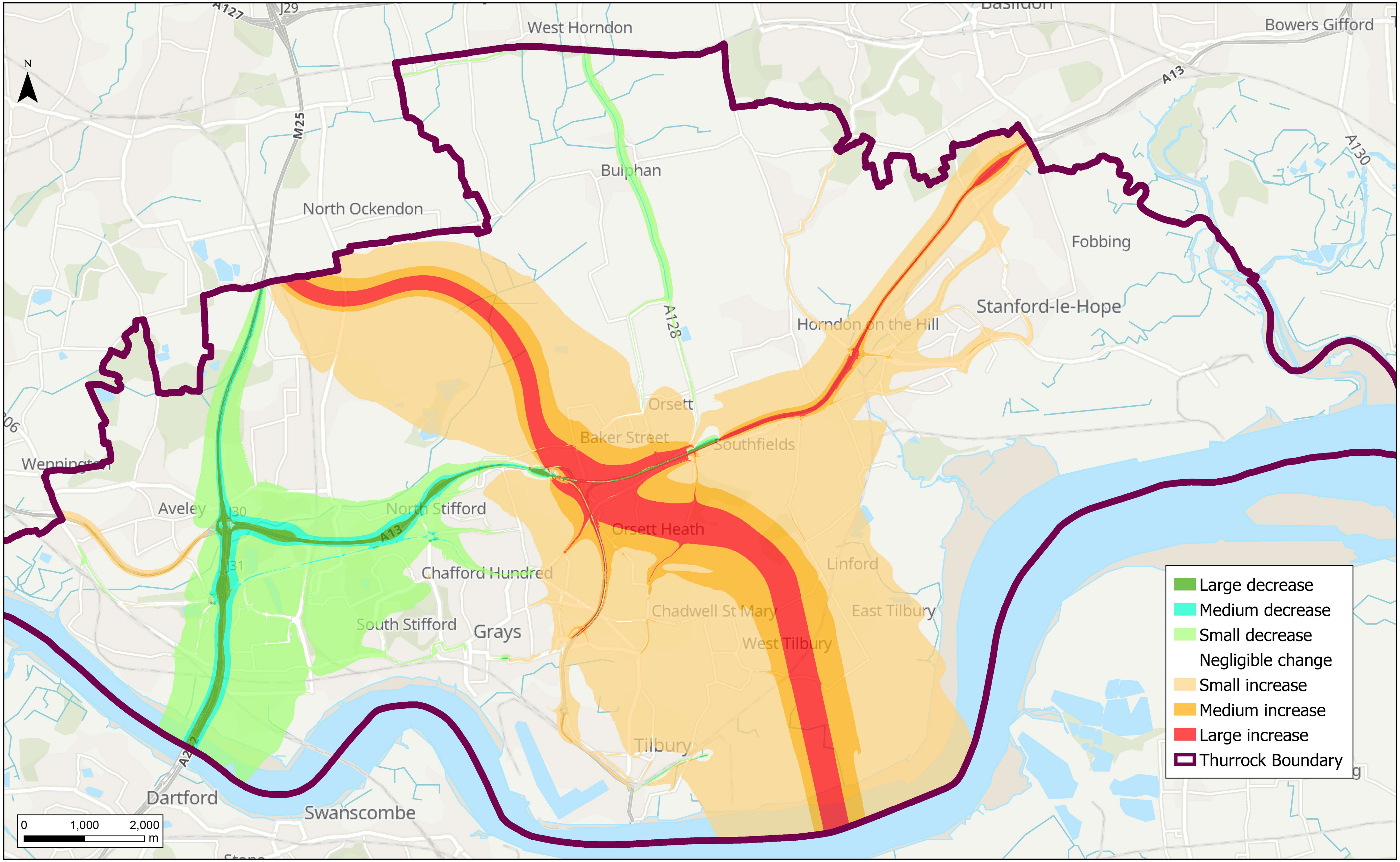
				Figure 02	Rev A



Stantec	Client THURROCK COUNCIL	**Thurrock Council Air Quality Modelling** Predicted change in annual average NO_2 concentration.	GB Topographic: Contains OS data © Crown Copyright and database right 2020 Contains data from OS Zoomstack	1:80,000 @ A4	Date: 10/05/2022
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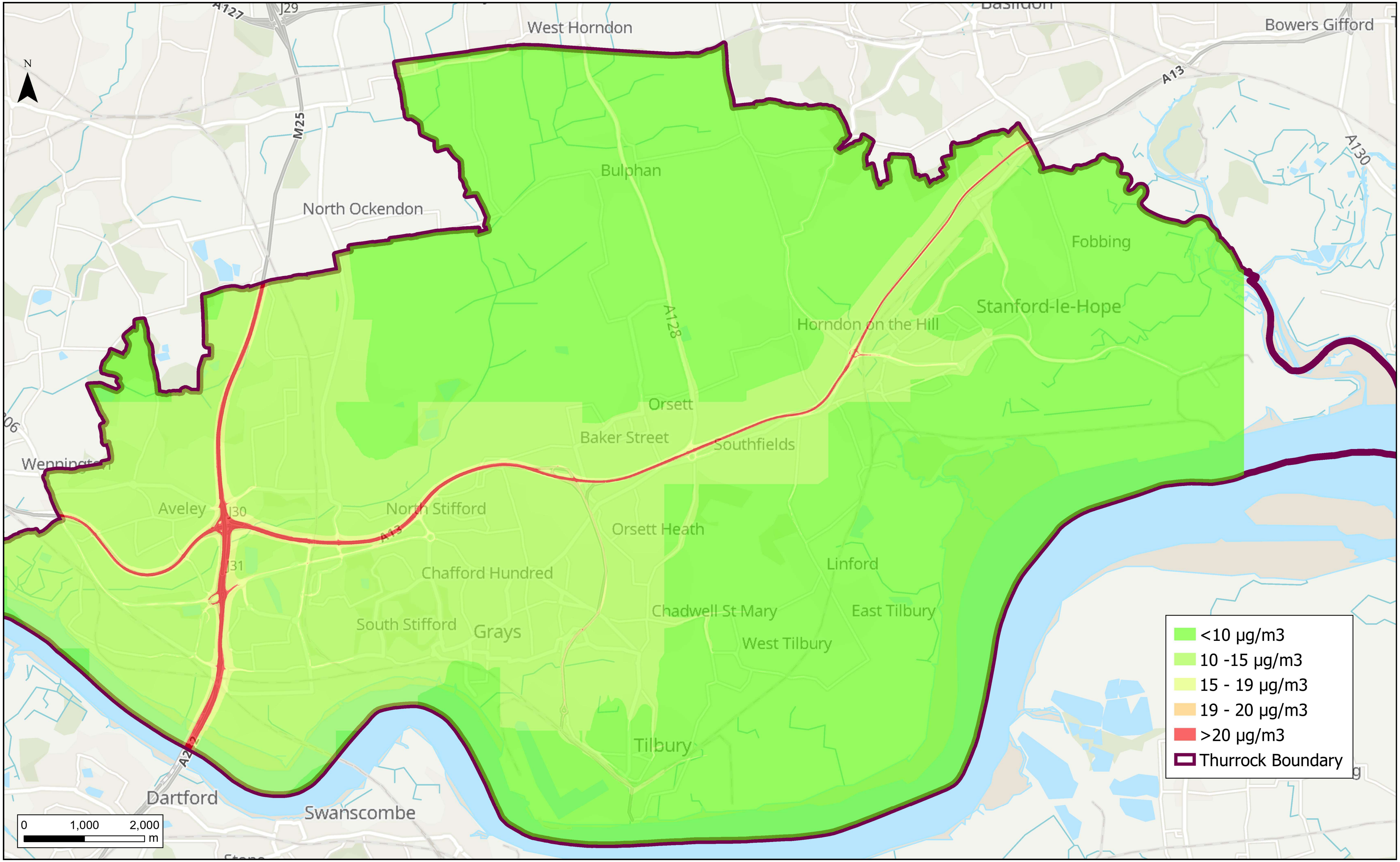
				Figure 03	Rev A



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Thurrock Council Air Quality Modelling

Predicted 'Do-minimum' annual average $PM_{2.5}$ concentration.

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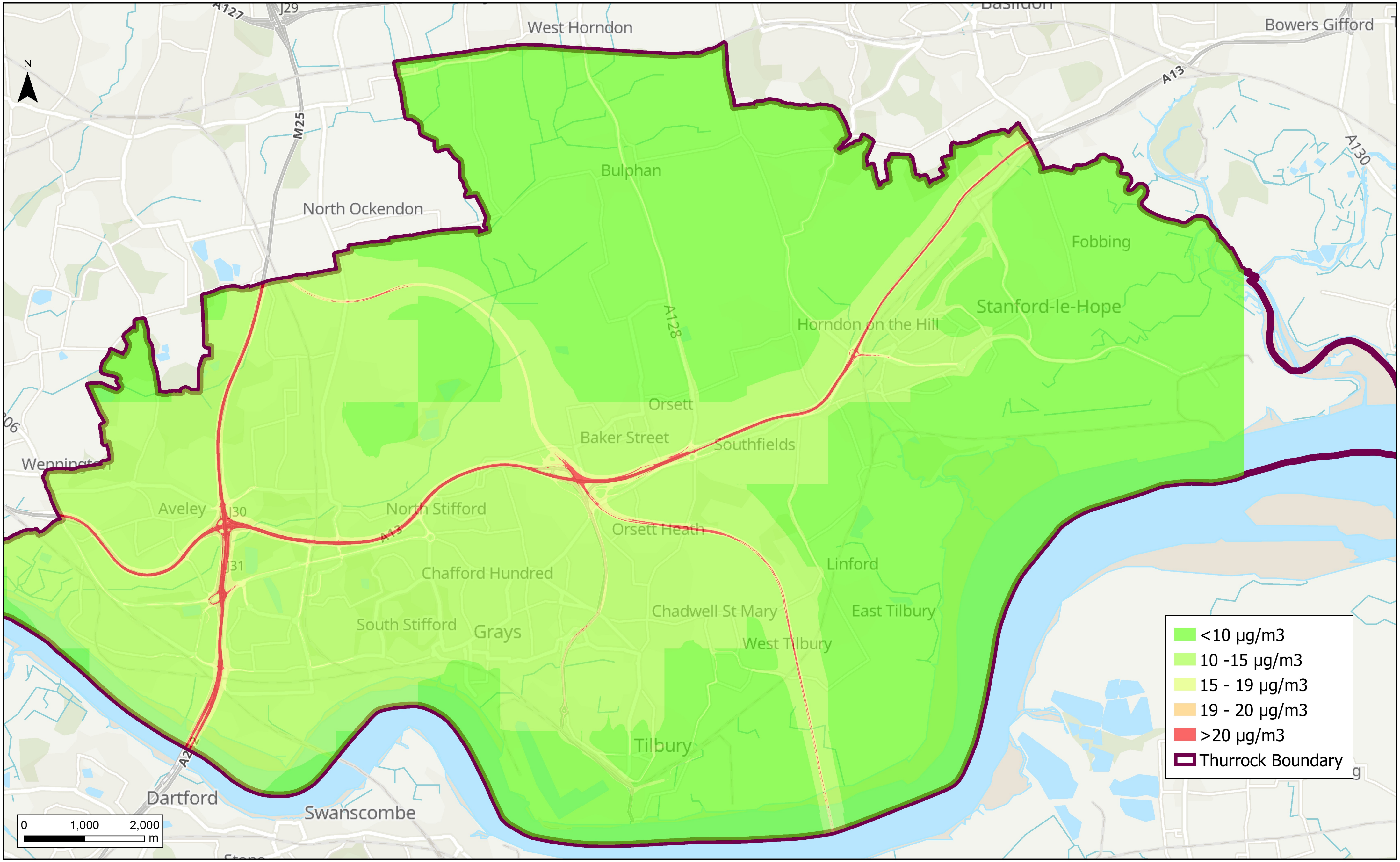

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Thurrock Council Air Quality Modelling

Predicted 'Do-something' (i.e. with LTC) annual average $PM_{2.5}$ concentration.

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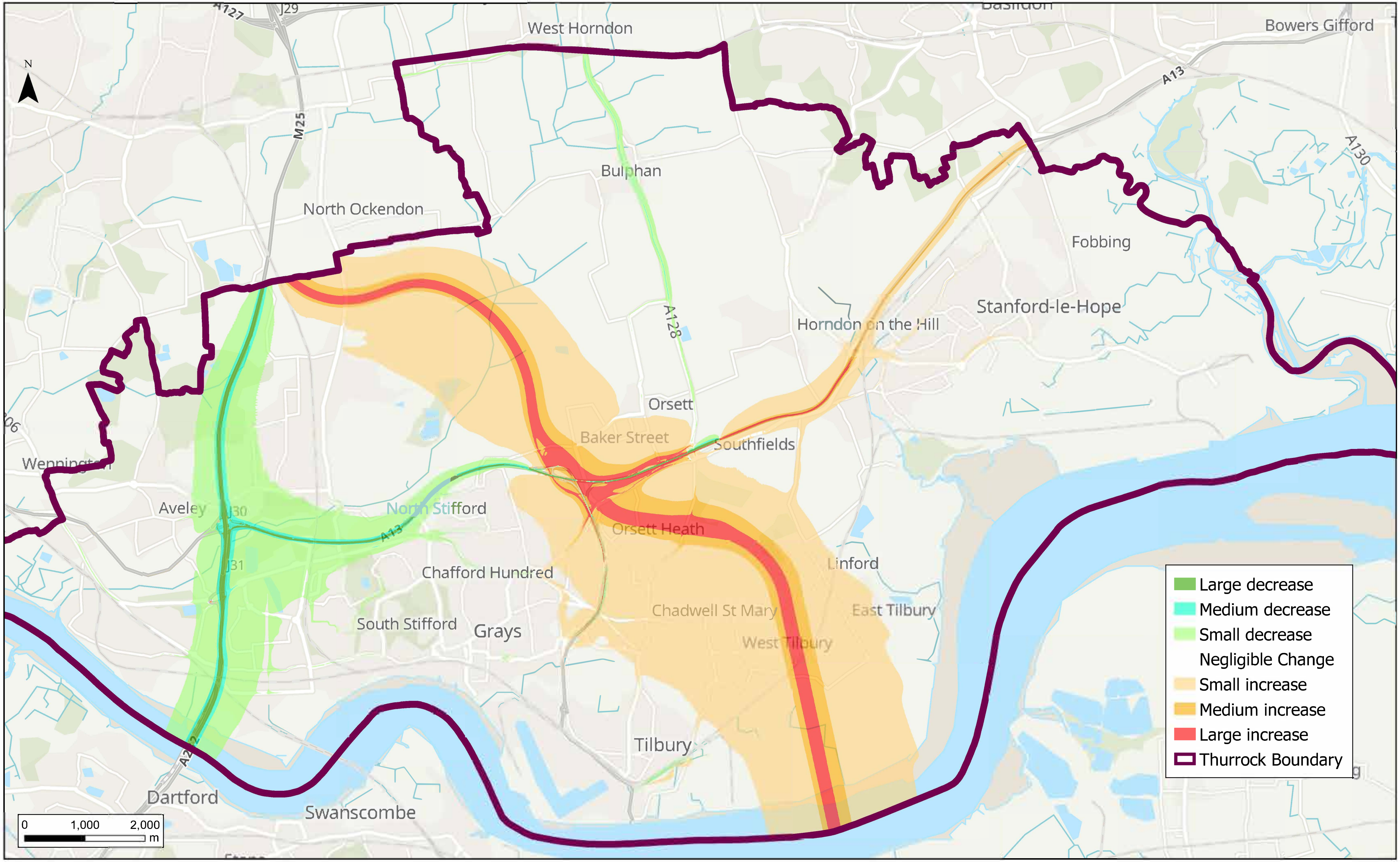
Figure 05

Rev A



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Predicted change in annual average $PM_{2.5}$ concentration.

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Appendix D – Annex 2 Noise Environment International Article

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Full length article

Spatial assessment of the attributable burden of disease due to transportation noise in England

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Environmental burden of disease
Environmental health

ABSTRACT

Background: Noise pollution from transportation is one of the leading contributors to the environmental disease burden in Europe. We provide a novel assessment of spatial variations of these health impacts within a country, using England as an example.

Methods: We estimated the burden of annoyance (highly annoyed), sleep disturbance (highly sleep disturbed), ischemic heart disease (IHD), stroke, and diabetes attributable to long-term transportation noise exposures in England for the adult population in 2018 down to local authority level (average adult population: 136,000). To derive estimates, we combined literature-informed exposure-response relationships, with population data on noise exposures, disease, and mortalities. Long-term average noise exposures from road, rail and aircraft were sourced from strategic noise mapping, with a lower exposure threshold of 50 dB (decibels) L_{den} and L_{night}.

Results: 40 %, 4.5 % and 4.8 % of adults in England were exposed to road, rail, and aircraft noise exceeding 50 dB L_{den}. We estimated close to a hundred thousand (~97,000) disability adjusted life years (DALY) lost due to road-traffic, ~13,000 from railway, and ~ 17,000 from aircraft noise. This excludes some noise-outcome pairs as there were too few studies available to provide robust exposure–response estimates. Annoyance and sleep disturbance accounted for the majority of the DALYs, followed by strokes, IHD, and diabetes. London, the South East, and North West regions had the greatest number of road-traffic DALYs lost, while 63 % of all aircraft noise DALYs were found in London. The strategic noise mapping did not include all roads, which may still have significant traffic flows. In sensitivity analyses using modelled noise from all roads in London, the DALYs were 1.1x to 2.2x higher.

Conclusion: Transportation noise exposures contribute to a significant and unequal environmental disease burden in England. Omitting minor roads from the noise exposure modelling leads to underestimation of the disease burden.

1. Introduction

Noise from road traffic, rail, and aviation transport affects millions of people in Europe (EEA, 2020) and are major contributors to the overall environmental disease burden (Hanninen et al., 2014; EEA, 2020). In England, the Environmental Noise Regulations 2006 (as amended) requires, on a five year cycle, the determination, through noise mapping, of exposure to environmental noise from major sources of road, rail and aircraft, and in urban areas (known as agglomerations) (HM Government, 2006). The Environmental Noise (England) Regulations 2006

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transpose the provisions of the European Commission's Environmental Noise Directive (END, 2002/49/EC) (HM Government, 2006). Strategic noise mapping undertaken in 2017 revealed that ~ 11.5 million people are exposed to noise above 55 dB L_{den} (day-evening-night noise) from roads (DEFRA, 2019a,c) and ~ 1.5 million from railways (DEFRA, 2019a,b), respectively. England is also a major hub for commercial aviation, with one of Europe's busiest airports (Heathrow) situated within the London urban area (UECNA, 2020). 55 dB L_{den} is a historic noise threshold used for reporting strategic noise mapping data in accordance with the END requirements. The health-based guidelines for noise published by the World Health Organization (WHO) Regional Office for Europe were 53 dB L_{den} for road and 54 dB L_{den} for railway noise, respectively (WHO 2018). The equivalent guideline for aircraft noise exposure is 45 dB L_{den}, however currently aircraft noise exposure is not routinely modelled down to this level in England.

Long-term exposure to environmental noise has been shown to adversely impact psychological and physiological health and wellbeing (Guski et al. 2017; Hegewald et al. 2020; WHO 2018). Noise exposure at night can disturb sleep patterns, with downstream short and long-term consequences on daytime cognitive alertness, memory consolidation, quality of life and health (Basner and McGuire 2018). Exposures during daytime and night-time are also associated with cardiometabolic diseases such as incident cases of ischemic heart disease (IHD), stroke, and diabetes, with the majority of the epidemiological evidence related to road-traffic sources (Roswall et al. 2021; Sakhvidi et al. 2018; van Kempen et al. 2018; Vienneau et al. 2019). Mechanistic studies point to sustained stress responses in the body and thus repeated stimulation of the sympathetic nervous and endocrine systems as etiological pathways (Eriksson and Pershagen 2018; Munzel et al. 2014; Munzel et al. 2021a; Munzel et al. 2021b). Furthermore, noise impaired sleep is thought to be an important pathway for adverse cardiometabolic impacts through elevation of blood pressure and stress hormones, and induced changes to appetite and glucose dysregulation (Basner and McGuire 2018; Eriksson and Pershagen 2018; Munzel et al. 2021a).

Comparative burden of disease assessments provide important input into health decision-making and planning processes, research prioritization, and funding (Ezzati et al. 2004). For such comparisons to be made across health outcomes, risk factors, and locations, standardized metrics are needed. The Disability Adjusted Life Year (DALY) is one such metric that combines the years of life lost due to premature death and years of life lived in less than full health as a result of disease (WHO 2020). Several studies have quantified the attributable DALYs lost, or other measures of health burden, due to transportation noise exposures, largely within Europe at the city, regional or country level (Eriksson et al., 2017; Hanninen et al., 2014; Hegewald et al., 2021; Khomenko et al., 2022; Mueller et al., 2017; Murphy and Faulkner, 2022; Sohrabi and Khreis, 2020; Stassen et al., 2008; EEA, 2020; WHO Regional Office for Europe, 2011). The European Environment Agency (EEA) estimated that among a total of 33 European countries in 2017, 453,000 DALYs were lost to annoyance (being highly annoyed), 437,000 to sleep disturbance (being highly sleep disturbed), and 156,000 to ischaemic heart disease (IHD), as a result of environmental noise exposures. These estimates do not highlight the unequal distribution of attributable disease burdens within countries, such as within England, as a result of geographic variations in noise exposures and underlying disease incidence and mortality.

Using a more granular and geographically flexible approach, our study quantified, compared, and mapped the burden of annoyance (highly annoyed), sleep disturbance (highly sleep disturbed), IHD, stroke, and diabetes attributable to transportation noise exposures within England at the national, regional, and local authority level. Specifically, we estimated Population Attributable Fractions (PAF), the number and percentage of the population highly annoyed and sleep disturbed, and DALYs lost, in the adult population in England (n = 42,738,500) in 2018 due to long-term average noise exposures ≥ 50 dB L_{den} and L_{night} derived from strategic noise maps of road, rail, and aircraft sources.

2. Material and Methods

2.1. Environmental burden of disease approach

We calculated the burden of disease attributable to noise exposures from road, rail, and civil airports in England at the national, regional, and local level using established epidemiological methodologies (Murray et al. 2003). To compare the burden of disease across noise sources and health outcomes, we used a standard metric, the Disability Adjusted Life Year (DALY). Key pieces of information needed for these calculations were:

- Noise exposures and population distributions
- Population data on disease occurrence, mortality rates, and life expectancy
- Disability weights (i.e., weighting factor that reflects the relative severity of a disease/health state)
- Exposure-response relationships (ERR) between health outcomes and noise exposures

Due to limitations in the availability of public data on exposure, demographic, and health at the spatial resolution required for our assessment, we made several assumptions on temporal stability, which are discussed in more detail in the following sections. Our estimates are based on population demographic and health data for the year 2018, and population exposure distributions based on noise mapping carried out for the year 2012.

2.2. Geography and population

We conducted the analysis in England and looked at trends at the national, regional, and local authority level. Geographic boundaries for nine regions and 314 Local Authority Districts (LAD) were obtained from the UK Office of National Statistics (licensed under the Open Government Licence v.3.0; contains OS data © Crown copyright and database right) (ONS Geography 2021) (Fig. 1). Regional populations ranged in size from ~ 2.5 to 9 million people, while LAD populations ranged in size from ~ 2.2 thousand to 1.1 million people. As the majority of the evidence of the health effects of noise is from cohort studies with adult subjects, we limited the analysis to the adult population normally resident in England (20 + years, n = 42,738,500) in 2018 (NOMIS 2019).

2.3. Road, railway, and aircraft traffic noise exposure data

We utilized the 2012 noise exposure datasets for road, rail, and aircraft sources generated by the 'Round 2 (II)' strategic noise mapping to fulfil the requirements of the Environmental Noise (England) Regulations 2006 (ENR) (HM Government, 2006). For road traffic, exposure was assessed via a single national exposure dataset produced by combining:

- exposure to major sources (defined as a trunk road, or a motorway, or a principal or classified road that has more than three million vehicle passages a year), for receptors outside agglomerations (DEFRA, 2014a), with
- exposure to all motorways and classified (A roads) roads, for receptors living within agglomerations.

The datasets also included noise exposure from:

- Major railways: railways that record>30,000 train passages per year.

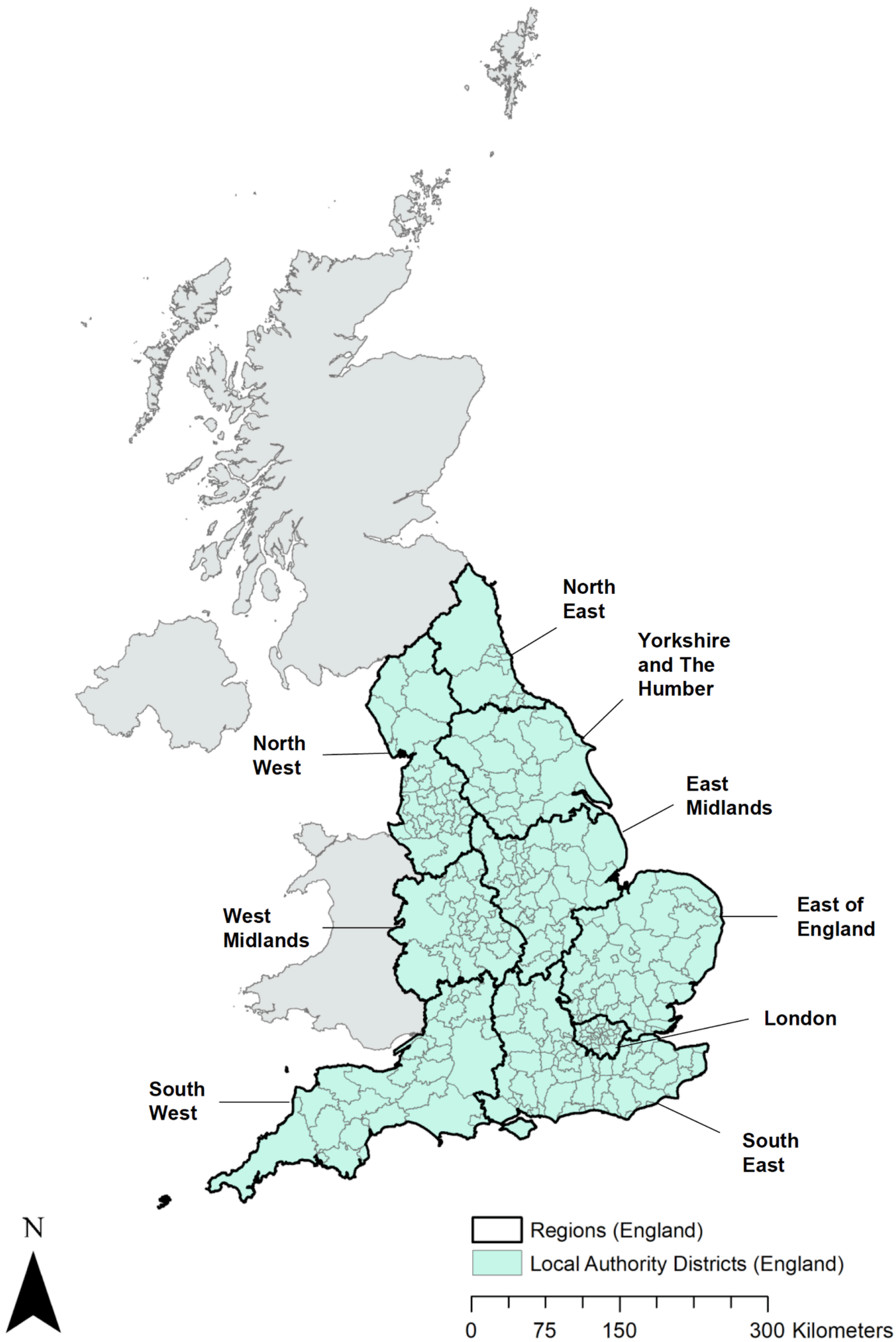

Fig. 1. Regions and Local Authority Districts (LADs) in England. Local Authority District boundaries (2020) were sourced from the Office for National Statistics Open Geography portalx (ONS Geography 2021) licensed under the Open Government Licence v.3.0 (contains OS data © Crown copyright and database right 2022). The geographic boundaries for the United Kingdom, England, and Regions within England were sourced from the Ordnance Survey website (Contains OS data © Crown copyright and database right 2022) (Ordnance Survey 2022).

- Major airports: civil airports, which record>50,000 movements per year (a movement being a take-off or a landing), excluding those purely for training purposes on light aircraft.

Modelled noise exposures are 'A' Frequency Weighted. The 'A' weighting is a standard weighting of the audible frequencies designed to reflect the response of the human ear to noise (weighted between 20 Hz and 20 kHz). $L_{den,}$ also known as the day-evening-night noise indicator, is the annual average A-weighted equivalent noise level (Leq) over a whole day, but with a penalty of + 10 dB for night-time noise (23:00–07:00) and + 5 dB for evening noise (19:00–23:00). L_{night} is the annual average A-weighted equivalent noise level over the 8-hour night period of 23:00 to 07:00 h, also known as the night noise indicator (WHO 2018).

The 'Round 2 (II)' strategic noise maps of England and the associated residential population exposure assessment, were produced by Extrium environmental consultants under contract to the UK Department for Environment, Food & Rural Affairs (DEFRA). The 'Round 2 (II)' noise mapping data production and exposure assessment process is known to be based on input data which had a high degree of currency consistency. This included road and rail transport movements; Ordnance Survey topographic data, e.g., the locations of buildings, noise barriers and transport infrastructure; Ordnance Survey address data and the 2011 Census. Source specific exposures were modelled for the façade of a building that faces the nearest major noise source, which is theoretically the point of maximum exposure. This was undertaken by intersecting residential building data features (vector polygons) with noise levels which are structured in 10 m × 10 m grid datasets. The maximum noise grid level is then attributed to each residential building and ultimately to the people assigned to each residential building following the residential population distribution process. Population level exposures were then calculated for the 2011 Census Output Area (COA) communities, which are typically formed of 300 residents living in 125 households. Residential buildings were identified from postal addresses, categorised into 1 dBA increments of L_{den} and L_{night} noise levels ranging from 50 dB to ≥ 80 dB, and then population counts were evenly assigned to the households. In 2011, there were 171,372 COAs in England that perfectly nested into the 314 Local Authority Districts operating in 2020. The COA population counts by exposure bands were then aggregated to form population exposure distributions (%) by LAD.

We then estimated LAD population exposures to noise in 2018 by applying the noise mapping exposure distributions to population counts recorded in 2018 by LAD. Doing this, we assumed that (i) the contribution and spatial distribution of major noise sources remained temporally stable between 2012 and 2018, and (ii) exposure distributions were the same for all age groups. These assumptions of uniform internal population dynamics were imposed by the decision to use the 2012 'Round 2 (II)' rather than the 2017 'Round 3 (III)' strategic noise maps of England. Furthermore, by using the 'Round 2 (II)' noise level data and the exposure assessment model, we had access to the underlying noise exposures in 1 dB bands and data starting consistently at 50 dB for all noise indicators. END exposure data in the public domain a) is in 5 dB bands, and b) starts from 55 dB L_{den}. Whilst road and railway noise exposure distributions are expected to be relatively stable over a period of six years, our assumptions may have introduced uncertainties for the aircraft noise estimates, due to growth in the aviation sector (Department for Transport 2022), replacement of older noisier aircraft types with more modern quieter ones (Civil Aviation Authority 2021c) and potential changes to flightpaths (Civil Aviation Authority 2021b) during this period. The population data was sourced from NOMIS, the Official Labour Market Statistics service provided by the Office for National Statistics (ONS) (NOMIS 2019) and was based on adult persons normally resident in English Local Authority Districts in 2018.

2.4. Health outcomes

Following a systematic review of reviews (Chen et al. 2022), we selected health outcomes based on the strength of the epidemiological and mechanistic evidence (Eriksson and Pershagen 2018) of the health effects of noise. The health outcomes included: annoyance (highly annoyed (HA)) (Fenech et al. 2022; Guski et al. 2017), sleep disturbance (highly sleep disturbed (HSD)) (Basner and McGuire 2018; Smith et al. 2022), IHD (also referred to as coronary heart disease) (van Kempen et al. 2018; Vienneau et al. 2019), stroke (Roswall et al. 2021; van Kempen et al. 2018), and diabetes (Sakhvidi et al. 2018; Vienneau et al. 2019). We did not quantify cognitive impairment in children (Clark et al. 2021) as our study focused on the burden of disease among adults.

2.4.1. Primary exposure response relationships (ERRs)

In cases where there were more than one recent systematic review/*meta*-analysis proposing an ERR for a health outcome and exposure pair, we considered the chronology of the publication and data (preference given to reviews with the most up-to-date evidence) and whether the evidence came from a published peer-reviewed paper versus a conference paper (preference given to peer-reviewed publications). We also only considered relative risks (RR) that were statistically significant (95 % confidence intervals around the central estimate did not cross zero) and which were associated with a specific traffic source (road, rail, or aircraft as opposed to 'total noise'). As such we selected ERRs for IHD and stroke from the WHO-commissioned systematic review by van Kempen et al (van Kempen et al. 2018) which considered studies published up until 2015, and for diabetes by Sakhvidi et al (Sakhvidi et al. 2018) which considered studies published up until 2017. It should be noted that the ERR for incident stroke in van Kempen et al is derived from only one large cohort study, and so as described in Section 2.4.2, we conduct a sensitivity analysis with an alternative ERR derived from pooling multiple cohorts from Scandinavian countries (Roswall et al. 2021). We used ERRs for sleep disturbance (HSD) from Smith et al (Smith et al. 2022), which updated the WHO-commissioned systematic review (Basner and McGuire 2018) by including studies published until 2021. Finally we used ERRs for annoyance (HA) from the WHO-commissioned systematic review by Guski et al (Guski et al. 2017) which included studies published up until 2014. Within their review, Guski et al identified a subgroup of studies from Asia and the Alpine valleys in Austria with several study characteristics that could influence the results, including the range of noise exposure levels, housing characteristics linked to ventilation and air conditioning, geographical terrain, and the annoyance scale cut-offs used to determine the highly annoyed category. As such, they proposed two ERRs for road-traffic noise: one using the full dataset of studies and another with Asian and Alpine studies excluded. For our study based in England, we used the ERR which excluded Asian and Alpine studies. ERRs can be found in Table 1.

Due to a lack of evidence and/or statistically significant relative risk estimates, we did not calculate attributable burden of disease for railway noise and IHD, stroke, and diabetes, as well as aircraft noise and stroke and diabetes. Though as the epidemiological evidence base develops and strengthens, it is possible that significant and robust relative risk estimates for these exposure-outcome pairs may be identified in the future.

2.4.2. Secondary ERRs for sensitivity analyses

We conducted a series of sensitivity analyses with alternative ERRs which either reflected an update to the evidence base or the consideration of study context and inclusion in ERRs. As such, we conducted the following sensitivity analyses:

- IHD and road-traffic noise: ERR proposed by Vienneau et al (Vienneau et al. 2019) of 1.02 [95 % CI: 1.00–1.04] RR increase per 10 dB L_{den}. Results were published in a conference paper and included studies published up until 2019.
- Stroke and road-traffic noise: ERR proposed by Roswall et al (Roswall et al. 2021) of 1.06 [95 % CI 1.03–1.08] Hazard Ratio (HR) per 10 dB L_{den}. Results were based on pooling data from nine large-scale Scandinavian cohorts (135,951 participants).
- Annoyance (HA) and road-traffic noise (a): ERR proposed by Guski et al (Guski et al. 2017) which used the full WHO dataset and included studies published up until 2014.
- Annoyance (HA) and road-traffic (b) and railway noise: ERRs proposed by Fenech et al (Fenech et al. 2022) which updates the evidence from the WHO-commissioned systematic reviews by including studies published up until 2022. Results are published in a conference paper.

2.5. Population attributable fractions (PAF)

To estimate the proportion of all new cases of disease, or other adverse condition, in a population that is attributable to a specific exposure (Calentano et al. 2019; Murray et al. 2003) we calculate Population Attributable Fractions (PAF). To calculate a PAF, the exposure distribution within a population, as well the relative risk of disease due to exposure, must be known. Therefore, we used Equation 1 to calculate PAFs for IHD, stroke, and diabetes due to noise exposure for the adult population within each LAD in 2018:

$$\text{PAF} = \frac{\sum_{i=1}^{n} p_i \times (RR_i - 1)}{\sum_{i=1}^{n} p_i \times (RR_i - 1) + 1}$$

Where i represents a noise level in 1 dB increments; n is the total number of noise levels within the defined valid range for the PAF calculation; p_i represents the proportion of the population exposed to

Table 1
Parameters used to estimate the burden of disease attributable to transportation noise for the adult population in England in 2018.

	Health outcome	Noise metric	ERR source	ERR function/relative risk estimate [95 % confidence interval]	ERR lower	ERR upper	Disability weight [95 % confidence interval *if available*]
Road	Highly annoyed	L_{den}	(Guski et al. 2017) *	$\%HA = 116.4304–4.7342 \times L_{den} + 0.0497 \times L_{den}^2$ *Excluding Asian and Alpine studies*	40 dB	80 dB	0.02 (WHO Regional Office for Europe 2011)
	Highly sleep disturbed	L_{night}	(Smith et al. 2022) **	$\%HSD = 31.18323 - 1.47351 \times L_{night} + 0.01851 \times L_{night}^2$	40 dB	65 dB	0.07 (WHO 2009)
	Ischemic heart disease	L_{den}	(van Kempen et al. 2018)	1.08 [1.01–1.15] per 10 dB	53 dB	80 dB	0.405 (WHO 2018)
	Stroke	L_{den}	(van Kempen et al. 2018)	1.14 [1.03–1.25] per 10 dB	50 dB	70 dB	0.522 [0.377–0.707] (Salomon et al. 2015)
	Diabetes mellitus	L_{den}	(Sakhvidi et al. 2018)	1.07 [1.02–1.12] per 5 dB	50 dB	80 dB	0.049 [0.031–0.072] (Global Burden of Disease Collaborators 2017)
Railway	Highly annoyed	L_{den}	(Guski et al. 2017)	$\%HA = 38.1596 - 2.05538 \times L_{den} + 0.0285 \times L_{den}^2$	40 dB	85 dB	0.02 (WHO Regional Office for Europe 2011)
	Highly sleep disturbed	L_{night}	(Smith et al. 2022) **	$\%HSD = 63.56140 - 3.00711 \times L_{night} + 0.03717 \times L_{night}^2$	40 dB	65 dB	0.07 (WHO 2009)
Aircraft	Highly annoyed	L_{den}	(Guski et al. 2017)	$\%HA = -50.9693 + 1.0168 \times L_{den} + 0.0072 \times L_{den}^2$	40 dB	75 dB	0.02 (WHO Regional Office for Europe 2011)
	Highly sleep disturbed	L_{night}	(Smith et al. 2022) **	$\%HSD = 17.07421 - 1.12624 \times L_{night} + 0.02502 \times L_{night}^2$	40 dB	65 dB	0.07 (WHO 2009)
	Ischemic heart disease	L_{den}	(van Kempen et al. 2018)	1.09 [1.04–1.15] per 10 dBA	47 dB	75 dB	0.405 (WHO 2018)

ERR: Exposure response relationship; ERR lower: Lowest noise level at which the ERR is considered valid; ERR upper: Highest noise level above which the risk stays constant.
*WHO commissioned systematic review derived two ERR curves for highly annoyed due to road-traffic noise exposure. One curve utilizing the full WHO dataset and another excluded Asian and Alpine studies (Guski et al. 2017).
**Smith et al presented multiple curves for HSD. We used the 'combined estimate' where noise was explicitly mentioned in the question (Smith et al. 2022).

noise level *i*; and RR_i is the relative risk increase in the health outcome at noise level *i*. As relative risks in the noise epidemiological literature are often represented as 10 dB or 5 dB increment increases in L_{den} or L_{night}, we first scaled our literature informed relative risks (RR) to 1 dB noise increases assuming a linear relationship. In the absence of a consensus for the theoretical minimum risk exposure level associated with cardiometabolic diseases, for each exposure-outcome pair we assigned relative risk increases starting from a lower noise threshold level based on the noise ranges reported within each review/*meta*-analysis, or from the information contained in the individual studies within each review (Table 1). This approach ensured that we do not extrapolate relationships outside their validity range. For IHD, we assigned the lower ERR threshold to reflect the weighted average of the lowest noise levels measured in the studies, based on WHO guidance (WHO 2018). For the stroke ERR, 50 dB L_{den} reflects the noise level at the bottom 5th percentile (rounded from 49 to 50 dB L_{den}) within the cohort study (Sorensen et al. 2011; Sorensen et al. 2014) used in the WHO commissioned systematic review (van Kempen et al. 2018). For the diabetes ERR, 50 dB L_{den} reflects the noise level at the bottom 5th percentile (rounded from 49 to 50 dB L_{den}) within the study that had the majority weight (Sorensen et al. 2013) in the *meta*-analysis by (Sakhvidi et al. 2018). If the assigned lower ERR threshold started below 50 dB L_{den} or L_{night} (i.e., the noise level cut-off of our exposure data (see Section 2.3)), then the burden of disease attributable to exposures between that level and 49 dB would not be captured in the results. Furthermore, our noise exposure data had an upper threshold of 80 dB L_{den} and L_{night}; if actual exposures exceeded this limit, they were capped at 80 dB.

2.6. Percentage of the population highly annoyed and sleep disturbed

The percentage of the population highly annoyed and highly sleep disturbed within LADs was estimated directly from the quadratic exposure–response function equations provided by Smith et al and Guski et al (Guski et al. 2017; Smith et al. 2022). We calculated the number of highly annoyed and highly sleep disturbed adults by multiplying the number of adults within each 1 dB noise band above 50 dB L_{den} (HA) and L_{night} (HSD) by the percentage of highly annoyed and highly sleep disturbed individuals at the corresponding noise level.

2.7. Burden of disease

Our main measure of disease burden was the Disability Adjusted Life Year (DALY) (Equation 2). The DALY simultaneously considers the reduced health state due to disability before death (Years of Life Lived with Disability (YLD)) and the decline in life expectancy due to death (Years of Life Lost (YLL)). We estimated DALYs for LADs represented as total DALYs as well as DALY rates per 100,000 people to adjust for local population sizes.

$DALY = YLL + YLD$ (2).

We calculated YLL from IHD, stroke, and diabetes separately for males and females and by 5-year age bands within each LAD by multiplying the number of disease-specific mortalities by the life expectancy at age of death. We then aggregated the age and gender specific estimates to achieve a single LAD-level estimate. We sourced a 3-year average life expectancy LAD dataset from the UK Office for National Statistics for the years 2017–2019 (Office for National Statistics, 2020a). Life expectancy information was not available for the City of London. Therefore, regional life expectancies for London were used for this LAD. We estimated the number of disease-specific mortalities by combining data on annual mortality counts for the year 2018 (Office for National Statistics 2019) and disease-specific mortality fractions. Local records of mortalities by their primary cause were not publicly available beyond 2013; therefore, the fraction of mortalities for each health outcome were calculated over a 5-year period from 2009 to 13 (Office for National Statistics 2015). We defined the underlying cause of death using codes provided by the International Classification of Diseases Tenth Revision (ICD-10) (Appendix A).

We calculated the YLD from IHD, stroke, and diabetes within each LAD by multiplying the annual prevalence rates of disease by the disease-specific disability weights (Kim et al. 2022; WHO 2020). Similar to the WHO approach, we calculated prevalence-based YLD (WHO 2020) and therefore did not include information on average duration of disease into the calculation. Disease prevalence rates come from the Quality and Outcomes Framework reporting (NHS Digital 2022), which was sourced from the UK Office for Health Improvement and Disparities 'Fingertips' service (Office for Health Improvement & Disparities 2022). Prevalence records were obtained for 191 Clinical Commissioning Groups (CCG) in England, the highest spatial resolution of data that is currently available. CCG datasets were resampled to 314 LADs using population-weighted interpolation, this approach calculates average rates based on the intersections of population counts at postcode points and the two administrative boundaries. Prevalence data for the years 2011–2018 were additionally averaged to minimise the influence of any unusual temporal anomalies. Disability weights allow non-fatal disability as a result of disease/health states to be measured under a common unit, ranging between 0 (no disability) and 1 (full disability). The disability weights that we used for IHD (DW: 0.405), annoyance (DW: 0.02) and sleep disturbance (DW: 0.07) were published and used in WHO noise guidelines (WHO 2009; WHO Regional Office for Europe 2011), and additionally used to estimate noise attributable disease burdens in Europe (EEA, 2020) (Table 1). As used in the Global Burden of Disease studies, we used a DW for stroke relating to long-term consequences of severe cases (DW: 0.522) and for diabetes we used a DW relating to uncomplicated cases (0.049) (Global Burden of Disease Collaborators 2017; Salomon et al. 2015). Alternative DWs for each health outcome were considered and the impact on DALYs was quantified in sensitivity analyses (National Institute for Public Health and the Environment (RIVM) 2018; Salomon et al. 2015) (Appendix A).

2.8. Noise attributable burden of disease

As a final step, the noise attributable DALYs were estimated by multiplying the PAF (i.e., disease attribution to noise) by the DALY (i.e., disease burden). The noise attributable DALYs were estimated with respect to each noise source (road, railway, aircraft) and health outcome separately at the LAD level. The calculation procedure for annoyance and sleep disturbance followed a slightly different process, whereby the number of adults highly annoyance and sleep disturbed were multiplied by the corresponding disability weight to estimate the attributable DALYs. For annoyance and sleep disturbance, only the impacts on morbidity contribute to the DALY. We estimated attributable DALYs at the LAD level, and then aggregated up to the regional and national levels to provide results at varying geographical scales.

2.9. Sensitivity analysis of alternative road-traffic noise exposure distributions: Case-study applied to the London region

Our main analysis quantified the attributable burden of disease for the adult population in England exposed above 50 dB L_{den} and L_{night} from road, railway, and aviation (commercial airports) sources, based on noise modelling conducted to fulfil the ENR strategic noise mapping requirements. However, we are likely under-estimating the attributable burden of disease due to the (i) restricted exposure range (>50 dB) and because (ii) exposures from "minor" roads (i.e., collector, residential and local access roads) are not counted. Many collector roads can still carry significant volumes of traffic, meaning that a) dwellings exposed to arterial and busy collector roads will experience higher exposure than that indicated by strategic noise mapping, and b) dwellings only exposed to collector and local roads could have exposures within the exposure range (50 – 80 + dB) but may be excluded from strategic noise mapping estimates. Furthermore, the 50 dB threshold does not account for low-level exposures for some of the population living in relatively quiet areas. Therefore, we conducted a sensitivity analysis where the attributable burden of disease was estimated with an alternate exposure distribution derived from all road sources (*CNOSSOS-all roads* model) and with a wider exposure range, starting at 40 dB L_{den} and 35 dB L_{night}. We conducted this analysis for the London region only, due to the availability of data. The exposure estimates were created by the Centre for Environmental Health and Sustainability (CEHS) at the University of Leicester, using a road-transport noise model for 2013 in accordance to the European Commissions 'Common framework for noise assessment' (CNOSSOS) (Kephalopoulos et al. 2012).

The CNOSSOS noise propagation algorithms were implemented in PostgreSQL via the PostGIS v2.1 extension, following the protocol described by Gulliver et al. (Gulliver et al. 2015). Annual Average Daily Traffic (AADT) counts and traffic speeds are considered along with information relating to the surface roughness of land cover, building heights, wind profiles and annual average temperatures. Roads are divided into 10 m segments and ray paths are drawn to the receptor locations accounting for the angle of view, source distance, and façade reflections. The AADT counts used here are available for the entire UK road-network in 2013 and were modelled by (Morley and Gulliver 2016). The noise contributions from major (motorways and a-roads) roads within 1 km and minor roads within 100 m of the receptor location are modelled as separate components. The estimated noise level at each receptor is the sum of sound propagation from every road in the network, including all public-accessible minor and local roads. The *CNOSSOS-all roads* model was run for 140,793 address locations in London, which were selected by assigning population-weighted postcode centroids to the nearest building. The 33 LADs in London typically contain 4,266 postcodes (SD = 1,355), which on average house 58 residents (SD = 44). Noise levels are modelled at the loudest façade, identified by calculating the AADT count of the nearest road inverse to the roads distance.

2.10. Software

We used Microsoft Excel workbooks (Version 2022) for the burden of disease calculations, the open-source statistical computing language and environment R (R Core Team 2022) for generating figures/plots, and ArcGIS software by Esri (Version 10.5.1) for mapping.

3. Results

3.1. Population noise exposures in England

40 % of the adult population in England were exposed to road-traffic noise that exceeded 50 dB L_{den}, however, there was a large degree of spatial variation across LAD (Fig. 2). 4.8 % of the population was exposed to aircraft noise above 50 dB L_{den}, though the spatial distribution was particularly skewed as 95 % of LADs had less than 20 % of populations exposed. The highest exposures to aircraft noise from major airports were found near London. Lastly, 4.5 % of the population in England were exposed to railway noise from mainlines that exceeded 50 dB L_{den}, though there was less variation between LADs (Range: 0 – 21 %), compared with road-traffic and aircraft exposures. There were similar variations and spatial distributions for night-time noise (Appendix A). We also found that 27 % and 2.9 % of the adult population were exposed to road and railway noise above the WHO guidelines levels of 53 dB L_{den} (road) and 54 dB L_{den} (railway), respectively. However, we could not estimate the % of the population exposed above the L_{night} guideline levels for all sources (road: 45 dB; rail: 45 dB; aircraft 40), or for L_{den} aircraft noise (45 dB), as the guideline levels fall below the lowest exposure threshold of our data (50 L_{den} and 50 L_{night} dB).

3.2. Noise attributable burden of disease in England

Road-traffic accounted for the majority of the DALYs lost in England in 2018, accounting for close to 97,000 DALYs/yr (Table 2). This was

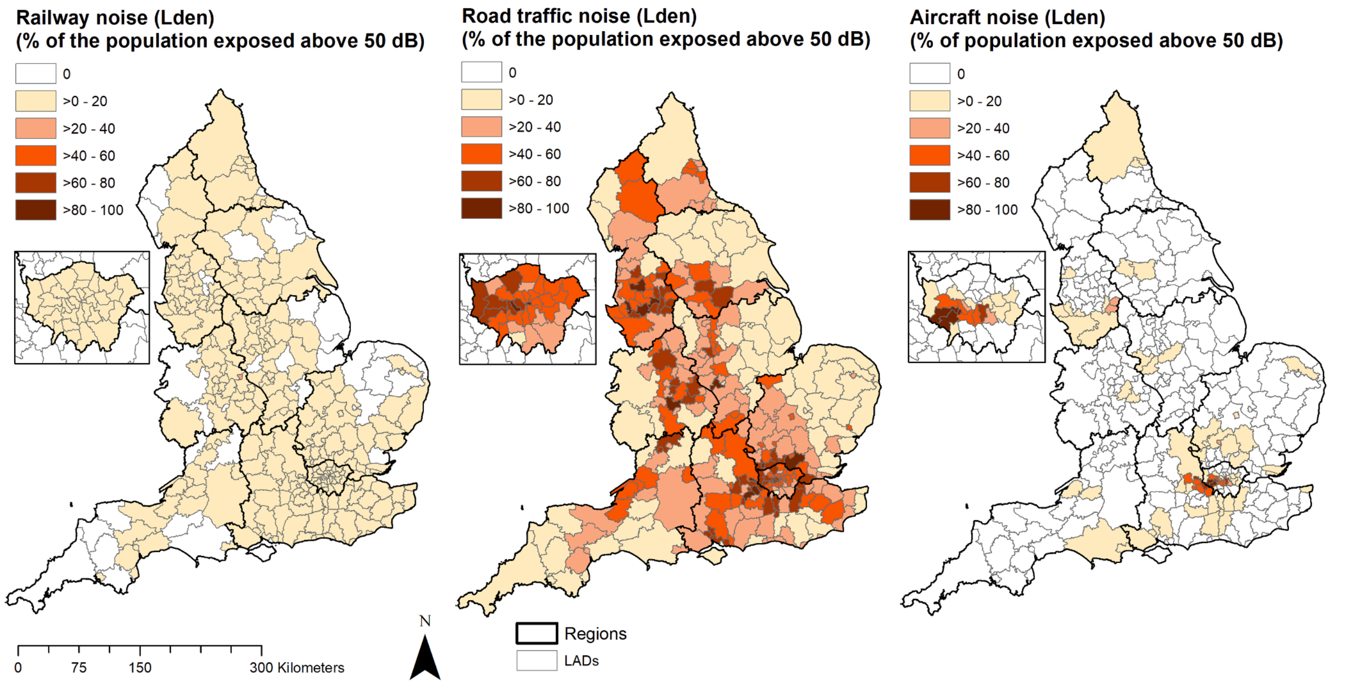

Fig. 2. Spatial variation in the percentage (%) of the population exposed to road-traffic, railway, and aircraft noise from major sources above 50 dB (L_{den}) across local Authority Districts (LADs) in England, based on strategic noise mapping carried out in 2012.

Table 2
Attributable burden of annoyance, sleep disturbance, ischemic heart disease, stroke, and diabetes mellitus due to road-traffic, railway, and aircraft noise exposures above 50 dB L_{den} and L_{night} in England. Estimates are for the adult population (20 +) in 2018.

		Road-traffic Central estimate [95 % confidence interval]*	Railway traffic Central estimate [95 % confidence interval]*	Aircraft traffic Central estimate [95 % confidence interval]*
Highly annoyed (HA)				
	% of population	3.9 %	0.7 %	1.2 %
	Number of people	1,662,157	295,766	524,321
	Total DALYs/yr	33,243	5,915	10,486
	DALYs per 100,000 people/yr	78	14	25
Highly sleep disturbed (HSD)				
	% of population	0.9 % [0.6 % – 1.2 %]	0.2 % [0.2 % – 0.3 %]	0.2 % [0.1 % – 0.2 %]
	Number of people	382,333 [236,040 – 521,004]	101,815 [66,788 – 127,132]	65,455 [47,155 – 83,625]
	Total DALYs/yr	26,763 [16,523 – 36,470]	7,127 [4,675 – 8,899]	4,582 [3,301 – 5,854
	DALYs per 100,000 people/yr	63 [39–85]	17 [11–21]	11 [8–14]
Ischemic heart disease (IHD)				
	PAF (%)	1.5 % [0.2 % – 2.7 %]	–	0.2 % [0.1 % – 0.4 %]
	Total DALYs/yr [95 % CI]	11,556 [1,427–21,942]	–	1,970 [876–3,273]
	DALYs per 100,000 people/yr [95 % CI]	27 [3–51]		5 [2–8]
Stroke				
	PAF (%)	3.8 % [0.8 % – 6.6 %]	–	–
	Total DALYs/yr [95 % CI]	18,592 [2,926–41,093]	–	–
	DALYs per 100,000 people/yr [95 % CI]	44 [7–96]		
Diabetes Mellitus				
	PAF (%)	4.2 % [1.2 % – 7.2 %]	–	–
	Total DALYs/yr [95 % CI]	6,686 [1,291–16,301]	–	–
	DALYs per 100,000 people/yr [95 % CI]	16 [3–38]		

Total DALYs/yr: Total number of Disability Adjusted Life Years lost per year; YLL: Years of Life Lost; YLD: Years of Life Lived with Disability; PAF (%): Population Attributable Fraction percentage (%); % of pop: Percentage of the population.

* The 95 % confidence intervals (CI) around the central burden of disease estimates of IHD, stroke, and diabetes were based on the combined uncertainty reported for the ERR functions, disease prevalence, disability weights, and life expectancy. While the 95 % CIs around the central estimate for sleep disturbance was based solely on the uncertainty estimate of the ERR function as the corresponding disability weight did not have a 95 % CI. We did not have uncertainty estimates for the ERRs or disability weights to be able to construct a 95 % confidence interval around the central burden of disease estimate for annoyance.

followed by aircraft (~17,000/yr) and then railway (~13,000/yr) noise exposures. Across health outcomes, annoyance accounted for the largest number of total DALYs/yr from road-traffic (33,243/yr) and aircraft (10,486/yr) exposures, while sleep disturbance accounted for the most from railway noise exposures (7,127/yr) (Table 2). From road-traffic noise exposures, we estimated 18,592 attributable DALYs/yr lost from strokes, 11,556 from IHDs, and 6,686 from diabetes. The attributable DALYs for stroke were higher than for IHD due to a combination of the magnitude of the relative risk estimates and the higher disability weight. The lower number of attributable DALYs for diabetes compared with IHD and stroke was driven by a lower disability weight (IHD: 0.405; stroke: 0.522; diabetes: 0.049) and annual mortality rates in the population (comparative statistics can be found at (British Heart Foundation, 2020).

When we tested the sensitivity of our estimates for a range of disability weights, we found that the total DALYs/yr for road-traffic noise varied between 7,040 – 11,556 for IHDs, 5,043 – 18,592 for strokes, and 2,763 – 13,003 for diabetes. By applying alternative disability weights for annoyance and sleep disturbance (National Institute for Public Health and the Environment (RIVM) 2018), the attributable DALYs/yr were reduced to 16,622 and 6,691, respectively. See Appendix A for more information.

3.3. Noise attributable burden of disease at the regional and local authority level

London had the greatest number of DALYs/yr lost attributable to road (~20,000/yr), railway (~5,000/yr) and aircraft (~11,000/yr) noise exposures compared with all other regions in England (Fig. 3 - upper panel). In fact, London had six times greater road-traffic DALYs compared to the region with the lowest levels (North East). The regional differences were particularly marked for aircraft noise exposures as the London region accounted for 63 % of all aircraft noise DALYs in England. The South East and North West of England also had relatively high number of total attributable DALYs/yr due to road, railway, and aircraft noise, while the North East had the lowest, compared with other regions.

When the influence of population size was removed from the road-traffic attributable DALYs by expressing them as rates per 100,00 people (Appendix A - Table A7), London still had the largest burden for annoyance (116 per 100,000 people) and sleep disturbance (101 per 100,000 people). However, the DALY rates for London were on par with the North West with respect to diabetes (London: 21, North West: 21) and in fact lower than the North West for stroke (London: 39, North West: 63) and IHD (London: 28, North West: 43). Not only did the absolute number of DALYs, and the rates, vary between regions, but so too did the proportional contributions of each health outcome to the total DALYs within each region. Notably, the proportional contribution of road-traffic IHD and stroke DALYs to the total DALYs within London was noticeably lower than the proportional contributions within other regions (Fig. 3 - lower panel).

There was a high-degree of variation in the DALY rates (per 100,000 people/yr) attributable to road-traffic noise exposure across LADs for annoyance (Interquartile range (IQR): 39–139), sleep disturbance (IQR: 29–109), IHD (IQR: 15–46), stroke (IQR: 28–75), and diabetes (IQR: 8–28). This variation also showed distinct spatial patterns across the country. We found the highest DALY rates clustered within and around London, the North West, and in the eastern part of the West Midlands (Fig. 4). For railway noise, the highest DALY rates were observed in local authorities within and around London, and those located along the major North-South railway routes in England (Fig. 5). For aircraft noise,

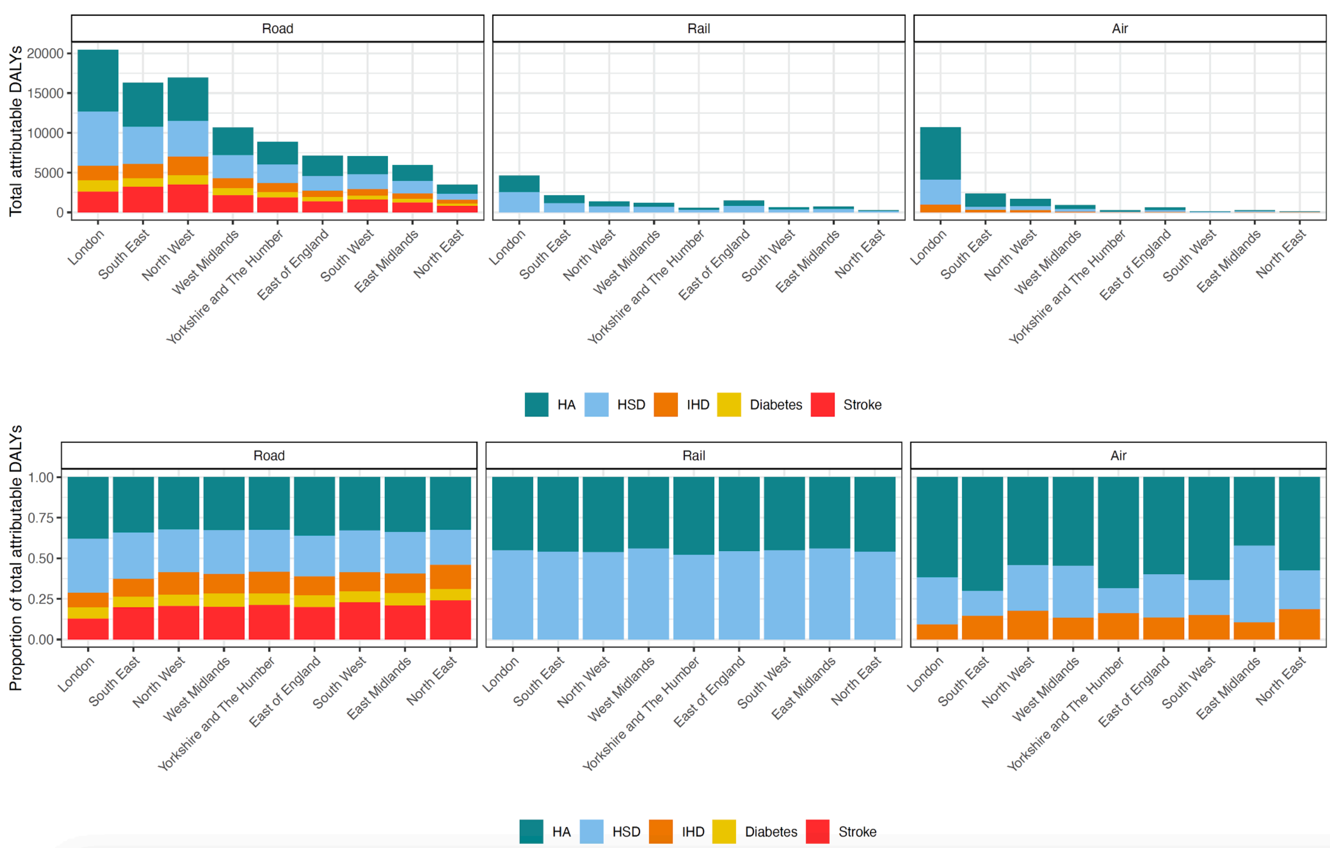

Fig. 3. Attributable Disability Adjusted Life Years (DALYs) lost due to road, railway, and aircraft traffic noise exposures above 50 dB across regions in England. Estimates are for the adult population (20 +) in 2018. Note that no robust ERRs were identified for railway noise and IHD, stroke, and diabetes, and for aircraft noise and stroke and diabetes.

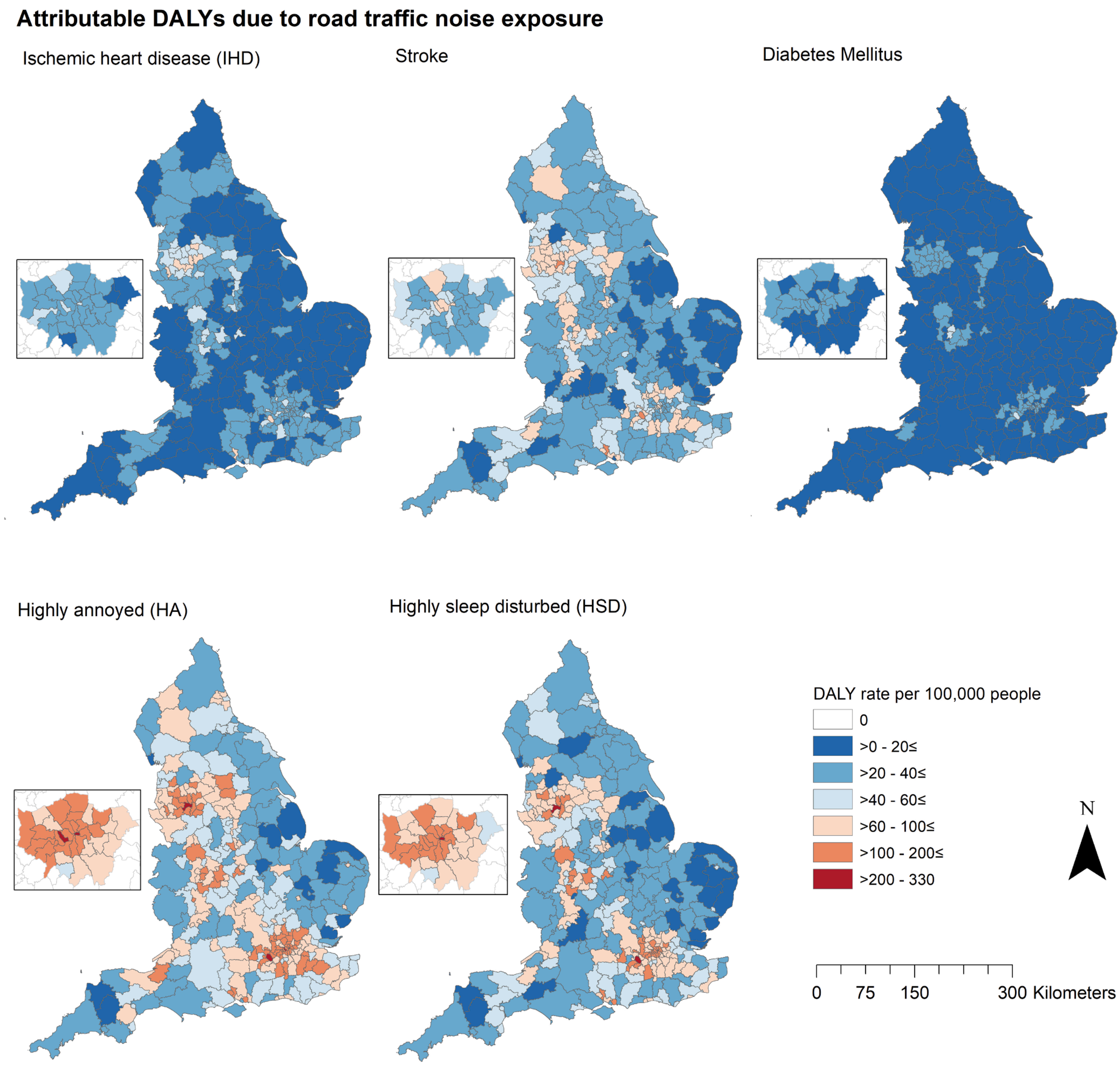

Fig. 4. Attributable Disability Adjusted Life Years (DALY) lost per 100,000 people/yr due to road-traffic noise exposures above 50 dB within Local Authority Districts (LAD) in England. Estimates are for the adult population (20 +) in 2018. Inset map is Greater London.

local authorities within and around London had by far the highest DALY rates. In particular Hounslow and Richmond near Heathrow airport (Fig. 6). DALYs shown by noise source, health outcome, and LAD are given in Appendix B, C and D.

3.4. *Sensitivity analysis of alternate exposure response relationships (ERR)*

The estimated burden of disease from IHDs and strokes attributable to road-traffic noise exposure was very minimally impacted by the application of alternative ERRs proposed by Roswall et al (stroke) and Vienneau et al (IHD) (Roswall et al. 2021; Vienneau et al. 2019) (details can be found in the Appendix A). We did, however, find that the attributable DALYs from annoyance (HA) was impacted by the choice of ERR. By applying alternative road-traffic ERRs, we found that the number of DALYs/yr were 1.4x (Guski et al. 2017) and 1.3x (Fenech et al. 2022) times higher than our main estimate. For railway noise, the alternative ERR for annoyance resulted in a smaller difference (1.15x times higher than our main estimate) (details in Appendix A).

3.5. *Sensitivity analysis of an alternative noise exposure distribution for London*

As a sensitivity analysis for the noise exposure modelling, the attributable burden of disease for London was estimated using an alternative noise exposure distribution calculated by the *CNOSSOS-All roads* model (see Section 2.9 for details on model). *CNOSSOS-All roads* uses estimates of traffic flows/speeds from all roads and provides data down to a lower exposure threshold (40 dB L_{den}, 35 dB L_{night}) compared with the England-wide *ENR strategic mapping*, as illustrated in Fig. 7.

Within London we estimated a higher number of attributable DALYs/yr with *CNOSSOS-All roads* compared with *ENR strategic mapping*. The relative increase in attributable DALYs/yr varied between health outcomes, ranging from 1.1x (IHD) to 2.2x (HSD) times higher (Table 3).

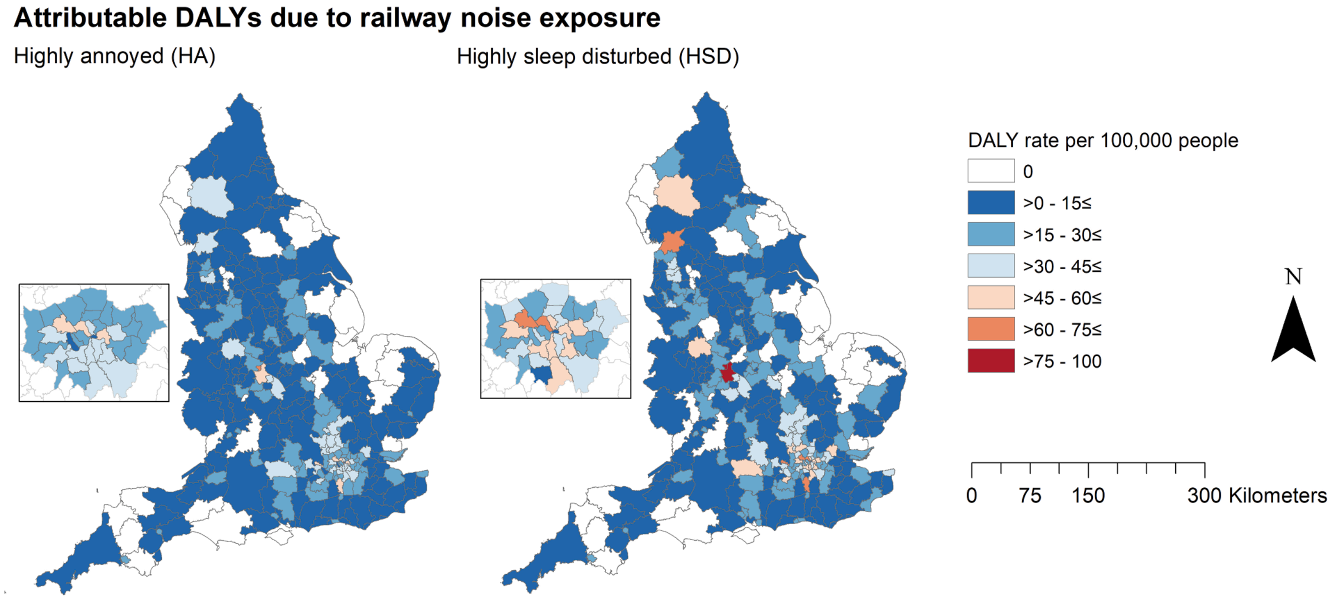

Fig. 5. Attributable Disability Adjusted Life Years (DALY) lost per 100,000 people/yr due to railway traffic noise exposures above 50 dB within Local Authority Districts (LAD) in England. Estimates are for the adult population (20 +) in 2018. Inset map is Greater London.

Sleep disturbance had the highest relative increase in DALYs compared with other health outcomes, likely because of the following factors: the lower ERR noise threshold (40 dB); the shape and shift in the L_{night} exposure distribution; and because the lower noise cut-off for L_{night} differed more between the models (35 to 50 dB L_{night}) compared with L_{den} (40 to 50 dB L_{den}). Based on these results, we presume that we are likely underestimating the road-traffic noise attributable burden of disease in England using exposure modelling from strategic noise mapping.

4. Discussion

While European studies have shown that environmental noise is a major contributor to the overall environmental disease burden (Hanninen et al., 2014; EEA, 2020; WHO Regional Office for Europe, 2011), detailed spatial assessments within countries are still lacking. Using recent and scientifically robust epidemiological evidence, we quantified, compared, and mapped the burden of annoyance, sleep disturbance, IHD, stroke, and diabetes attributable to long-term transportation noise exposures for adults in England at the national, regional, and local authority level. We estimated that overall, approximately a hundred thousand (~97,000) DALYs were lost in England in 2018 due to road-traffic, ~13,000 from railway, and ~ 17,000 from aircraft long-term averaged noise exposures above 50 dB. The magnitude of the attributable disease burden varied significantly across regions and LADs.

Compared to Global Burden of Disease study estimates of other types of pollution in England in 2018, our road noise attributable DALYs (~97,000 DALYs) were lower than for ambient $PM_{2.5}$ pollution (~269,000 DALYs), but higher than ambient ozone (~7,000 DALYs) and occupational noise (~29,000 DALYs) (Global Burden of Disease Collaborative Network 2020). However, we note these comparisons between risk factors should be made with caution due to differences in methodologies, assumptions, and the state of the evidence. Furthermore, whilst our estimates of the overall noise attributable burden of disease was significant, it may still be under-estimated in England due to limitations in modelling noise exposures designed specifically for strategic noise mapping (i.e., only including roads with very high traffic volumes and only modelling above 50 dB L_{den} and L_{night}). Our noise attributable DALY estimates at a national level for England were largely in line with similar estimates produced for the United Kingdom (England plus Scotland, Wales and Northern Ireland) by the European Environment Agency (EEA) using END mapping (EEA, 2020). The available public information from the EEA does not provide information on within-country variations, so comparisons at a regional and local district level were not possible.

One of the main drivers of spatial variations in noise attributable disease burdens is non-uniform exposure to noise. This is particularly evident for aviation, as we found that London had 63 % of all attributable aircraft noise DALYs in England. Between 2012 and 2019, air traffic at Heathrow, Gatwick and London City was approximately 40 % of the total UK air traffic (Department for Transport 2022). Variations in noise attributable disease burdens are also impacted by the distribution of the population. For example, the London region had the highest total attributable DALYs due to road-traffic noise, but accounting for population size, the rates of road-traffic attributable DALYs (per 100,000 people) in the North West region were on par with London. Furthermore, spatial variations in local disease burden (British Heart Foundation, 2021b; Steel et al. 2018) also contribute to spatial contrasts of noise attributable DALYs across England. This is evident in the fact that the proportional contribution of road-traffic IHD and stroke DALYs to the total DALYs within London was noticeably lower than the proportional contributions within other regions. This is because the underlying prevalence and mortality rates of IHD and stroke were generally lower in London compared with many other areas in England (comparative statistics can be found at (British Heart Foundation. 2021a,b) (also see Tables A8 and A9 in Appendix A), possibly because of the younger age demographic in London (median age 35.3yrs in London compared with 39.5 – 43.9 in other regions) (Office for National Statistics, 2020b).

The comparative environmental burden of disease concept facilitates comparisons across geographies, health outcomes, and risk factors, but comparisons should be made with care when the methodological assumptions and data inputs are not aligned across studies. For example, the epidemiological evidence on noise and health is developing and strengthening (Clark et al. 2020; Fenech et al. 2022; Persson Waye and van Kempen 2021; van Kamp et al. 2020), and therefore choices about which health outcomes to include, and their associated exposure response relationships will vary by study, and this can affect the comparability between studies. For example, a noise burden of disease study for Hesse, Germany (Hegewald et al. 2021) included depressive disorders based on relative risk estimates from a large study conducted around Frankfurt Airport (Seidler et al. 2017). We excluded depression

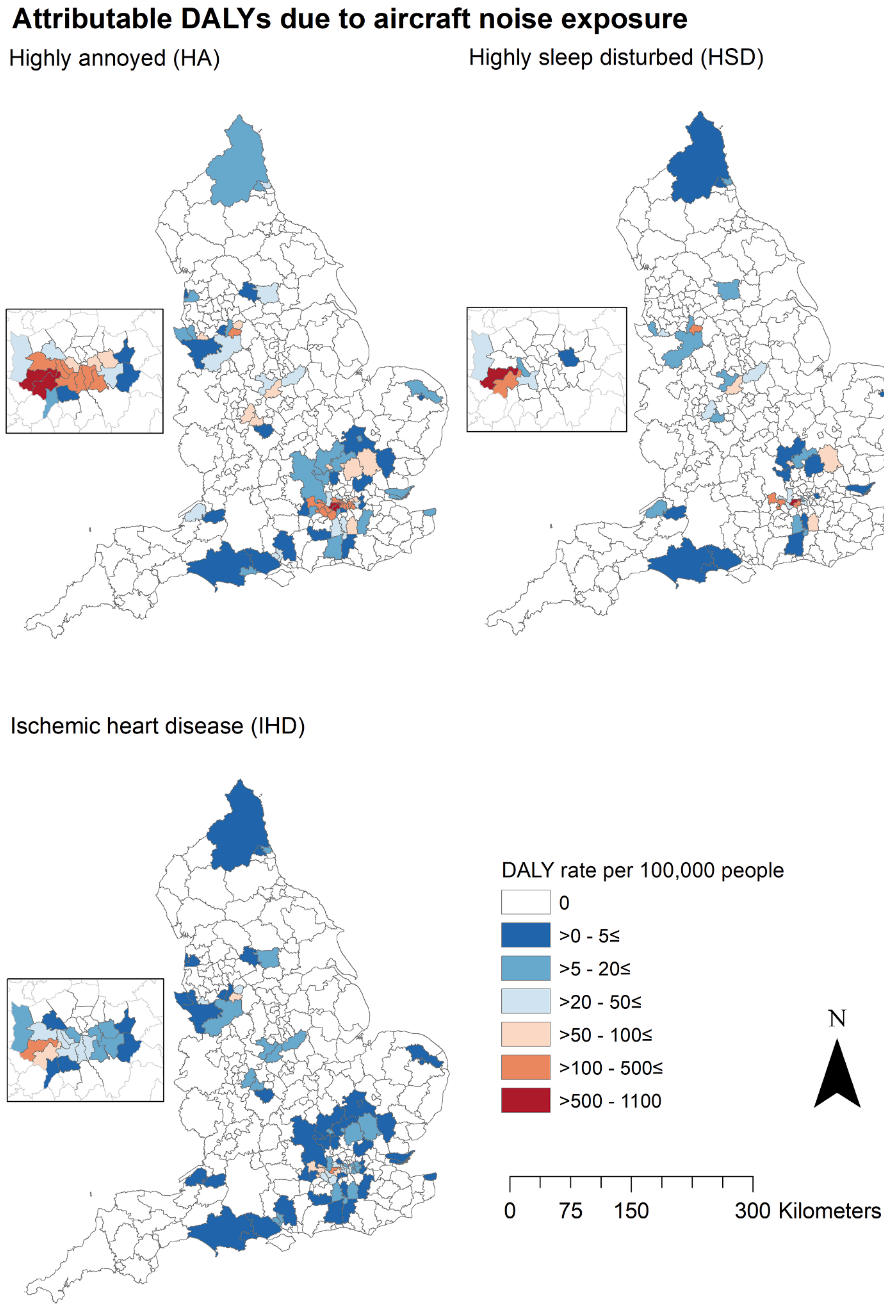

Fig. 6. Attributable Disability Adjusted Life Years (DALY) lost per 100,000/yr people due to aircraft traffic noise exposures above 50 dB within Local Authority Districts (LAD) in England. Estimates are for the adult population (20+) in 2018. Inset map is Greater London.

and anxiety from our assessment, however given that the evidence base is developing, we recommend that these health outcomes are kept under review for future updates. We also recommend that the shape of the exposure–response relationships for all health outcomes, and any potential threshold effects which would influence the choice of a counterfactual, also remain under continual review. There is limited but growing evidence of the shape of the exposure–response relationships and whether any potential thresholds of effect exist (Fu et al. 2022; Thacher et al. 2022; Vienneau et al. 2022). Another example is that the WHO-commissioned systematic review for annoyance from road traffic proposed two exposure response relationships (ERRs): a global one and one excluding Asian and Alpine studies. For our main analysis we used the latter, as did the aforementioned study in Hesse, Germany (Hegewald et al. 2020), whereas other European studies used the global ERR (Murphy and Faulkner, 2022; EEA, 2020). Within England, we found that the two curves led to a relative 1.4x difference in attributable road-traffic annoyance DALYs/yr. Another source for discrepancy is that some studies have applied ERRs from one noise source to another (e.g., using a road-traffic noise and IHD ERR for railway and aircraft exposures (EEA, 2020)) where information has been limited. We took a more conservative approach and only quantified the burden of disease if a source-specific statistically significant association was given in a review or *meta*-analysis of the epidemiological evidence. Our decision to use the same lower exposure cut-off of 50 dB for L_{den} and L_{night} from the strategic mapping means that the relative differences between the L_{den}-based YLD and the YLD due to sleep disturbance are not comparable to other assessments taking the more conventional approach of using a 5 dB lower cut-off for night-time noise. Lastly, similar to the WHO approach for DALY calculations (WHO 2020), we used disease prevalence estimates to derive YLD with respect to IHD, stroke, and diabetes,

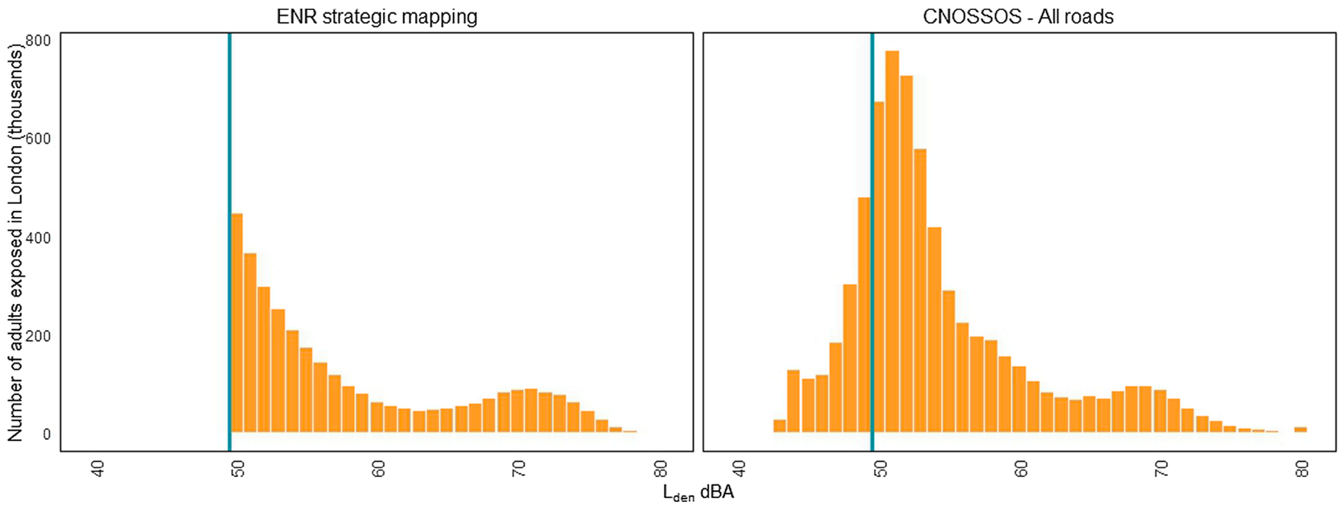

Fig. 7. Distribution of the number of adults exposed to road-traffic noise levels (Lden) in London based on ENR *strategic mapping* and *CNOSSOS – All roads* modelling. Vertical blue line indicates the threshold at the 50 dB L_{den} band. The numbers on the Y-axis are in units of thousands.

as incidence-based data was not available at the geographic resolution of our study. This could impact comparability with studies using the incidence-based calculation approach (Park et al. 2019; von der Lippe et al. 2020; Wagner et al. 2015).

For our main analysis we used exposure estimates from strategic noise mapping carried out to satisfy the requirements of the Environmental Noise (England) Regulations 2006 (as amended) (implementation of the European Environmental Noise Directive in England). Although a third round of strategic mapping was undertaken in 2017, we used noise mapping data from the second round (2012) for reasons already described in Section 2.3. For road and railway noise, the spatial distribution of exposure is relatively stable over time. The population within the > 55 dB L_{den} contour in 2017 was approximately 10 % higher than in 2012 for road and railway noise (DEFRA, 2014b; 2019). This increase is likely to have been due to a combination of changes in the population exposed and changes in noise emissions taking place between 2012 and 2017. Our analysis includes the former effect but not the latter. For aircraft noise, airports and air traffic control organisations in the UK sometimes carry out short-term trials of changes to the airspace structure (and hence flight paths) (Civil Aviation Authority 2023). More generally, the ongoing recovery of the aviation sector from the impacts due to Covid, and the upcoming large-scale changes to UK airspace (Civil Aviation Authority 2018) mean that any retrospective estimates of spatial variations in disease burden attributable to aviation noise should not be considered representative of future years, and regular assessment updates may be warranted (subject to availability of exposure data and up to date ERRs).

The availability of strategic noise mapping data greatly facilitates burden of disease assessments at national, regional and local levels; the downside is that not all roads, railways and airports may be included in the modelling, and there may be issues with the chosen exposure thresholds giving an incomplete picture of population exposure in the range where health effects are known to occur. As we showed with a sensitivity analysis for London, these two factors are likely to lead to under-estimation of the attributable disease burden, and demonstrates the importance of modelling all roads, and using a lower threshold than 50 dB. A similar result, but with even greater differences, was found for the Hessian population in Germany (Hegewald et al. 2021). Hegewald et al compared estimated DALYs due to road-traffic noise using exposures from END strategic mapping (exposures $\geq$ 55 dB) and 'PLUS-Mapping' which included all roads and noise levels as low as 40 dB. They found that DALYs estimated with PLUS-Mapping for cardiovascular diseases and depressive disorders were ~ 4.2x times higher than estimated with END strategic mapping. Although the absolute relative risk estimates for cardiometabolic health outcomes below 50 dB L_{den} can be low (van Kempen et al. 2018; WHO 2018), a large proportion of the population in England is likely to be exposed to road traffic noise within the 40 to 50 dB L_{den} range (DEFRA, 2000, 2013). Our sensitivity analysis in London used the CNOSSOS methodology (Kephalopoulos et al. 2012), whereas the strategic modelling used the UK's national calculation methodology (Calculation of Road Traffic Noise (CRTN) (Department of Transport et al. 1988)). We have not assessed the differences between the approaches in terms of exposure estimates, but we expect this to not significantly impact results when aggregated to our minimum unit of assessment (local authority district).

The sensitivity analyses of the ERRs showed that there was minimal difference in attributable DALYs for IHD and stroke when using the main and alternative ERRs. This was ascribed to the fact that while our main relative risk estimates were larger in magnitude than the alternatives, the lower thresholds for the alternative ERRs were set lower down on the exposure range, at 40 dB L_{den} (informed by the valid noise ranges in the included studies) resulting in similar DALY estimates. When setting lower ERR thresholds, it is important to consider the valid noise range from which each ERR was derived as it can influence the magnitude of the slope coefficient but also so as not to extrapolate relationships out of range of the data. Additionally, we conducted these comparisons for the population exposed to noise above > 50 dB (L_{den}), though if populations exposed to lower noise levels were to be included in the estimations (i.e., between 40 and 50 dB), then we might expect a widening of these differences. This may be particularly relevant for health impacts associated with night-time exposures, such as sleep disturbance, as a larger proportion of the population is exposed within the 40–50 dB L_{night} range. Lastly, the choice of ERR for annoyance had an impact on attributable DALYs, as evidenced by the differences observed when applying the road-traffic curves using the full WHO dataset versus curves excluding Asian and Alpine studies (Fenech et al. 2022; Guski et al. 2017). The WHO ENG guidelines recommended that data and ERRs derived in a local context should be used whenever possible (WHO 2018). As there has been no recent large-scale socio-acoustic study for road and railway noise in the UK (last one was in 1984) (Fenech et al. 2022), we used the WHO aggregate curve excluding Asian and Alpine studies for the main analysis, but we conducted sensitivity analyses with the full WHO curves (original (Guski et al. 2017) and updated curves (Fenech et al. 2022)) to provide additional estimates and to be transparent about the impacts of our methodological decisions. A socio-acoustic study on attitudes to aviation noise was carried out in England in 2014–15, however an ERR expressed in terms of an annual L_{den} is not yet published from this study (Civil Aviation Authority 2021a).

Table 3
Comparison of the impact of modelled noise exposure distributions on the road-traffic attributable burden of disease in London.

Health outcome		Road-traffic: strategic mapping *(ENR)** ≥50 dBA L_{den} ≥50 dBA L_{night}	Road-traffic: all roads *(CNOSSOS-All roads)*** ≥40 dBA L_{den} ≥35 dBA L_{night}	Relative increase (*CNOSSOS-All roads* compared with *ENR*)
Highly annoyed (HA)				
	% of population	5.8 %	8.0 %	
	Number of adults	388,389	534,467	
	Total DALYs/yr	7,768	10,689	1.38x higher
Highly sleep disturbed (HSD)				
	% of population	1.4 %	3.2 %	
	Number of adults	96,501	213,723	
	Total DALYs/yr	6,755	14,961	2.21x higher
Ischemic Heart Disease (IHD)				
	PAF	2.3 %	2.5 %	
	Total DALYs/yr	1,886	2,057	1.09x higher
Stroke				
	PAF	5.5 %	6.9 %	
	Total DALYs/yr	2,623	3,330	1.27x higher
Diabetes mellitus				
	PAF	5.9 %	7.3 %	
	Total DALYs/yr	1,381	1,704	1.23x higher

****ENR:*** Noise exposures derived from modelling to fulfil the ENR Round 2 strategic mapping requirements (main analysis).
****CNOSSOS-All roads:*** Noise exposures created by the Centre for Environmental Health and Sustainability at the University of Leicester in accordance with CNOSSOSS-EU modelling framework (Gulliver et al. 2015).

4.1. Strengths and limitations

While the majority of noise burden of disease studies derive estimates at a national or city-level, we generated high spatial resolution estimates across 314 Local Authority Districts across England. Our calculations took into account spatial variations in noise exposures, population distributions, and underlying disease prevalence and mortality across the country. We made estimates for a range of health outcomes, including annoyance, sleep disturbance, and ischemic heart disease. Given the strengthening of the epidemiological evidence, we also made estimates for strokes and diabetes, which have largely been uncharacterised in other studies (Eriksson et al., 2017; Khomenko et al., 2022; Murphy and Faulkner, 2022; EEA, 2020; WHO Regional Office for Europe, 2011). We utilised recent synthesised scientific evidence to derive our burden of disease estimates (Guski et al. 2017; Sakhvidi et al. 2018; Smith et al. 2022; van Kempen et al. 2018) and undertook a comprehensive set of sensitivity analyses to show the impact of using alternative exposure–response relationships and disability weights. Finally, we also showed that there can be a substantial impact (under-estimation) on the estimated attributable burden of disease from road-traffic noise when the exposure range is limited and when traffic flows on collector or local roads are not taken into account in the modelling.

Our study has several potential limitations. We quantified the attributable burden of disease assuming the population was exposed to transportation noise sources in isolation. There are still uncertainties on the actual health burden in the event of co-exposure to multiple sources of transport noise, and there may be a risk of double counting. While the evidence is still limited, the EEA estimated the potential for double counting for annoyance and sleep disturbance from the combined effects of multiple sources in the 2020 report (EEA, 2020) to be a maximum of 13 % for annoyance, 16 % for sleep disturbance, and negligible for IHD, particularly if the PAFs are below 0.10 (Houthuijs et al. 2018). Conversely, a limited but growing evidence base is investigating whether the combined risks from multiple sources may be larger than the sum of the individual risks (Seidler et al. 2019; Thacher et al. 2021; Thacher et al. 2022). As the evidence base develops, future work should consider how to incorporate these potential combined effects into noise burden of disease assessments. Due to these current uncertainties, we recommend that our burden of disease estimates should not be combined across noise sources.

Due to a lack of evidence and/or statistically significant relative risk estimates, we did not calculate attributable burdens of disease for railway noise and IHD, stroke, and diabetes, as well as aircraft noise and stroke and diabetes. Though as the epidemiological evidence base develops and strengthens, it is possible that significant relative risk estimates for these exposure-outcome pairs may be identified in the future. We did approximate the total DALYs associated with each transportation source by summing across health outcomes, though this approach did not take into account comorbidities in the DALY calculation. Given the uncertainty in these estimates, we reported the total DALYs to the nearest thousand.

We used yearly IHD, stroke, and diabetes disease prevalence data which were based on general practice (GP) reporting within the National Health Service (NHS) Quality and Outcomes Framework (QOF). The QOF prevalence data were based on the number of patients recorded as having disease on the practice register over the total practice list size (Office for Health Improvement & Disparities 2022). These data are limited however, as people who have not been formally diagnosed would not be included in the estimate and it is also possible that there are variations across practices in how conditions are diagnosed (though this is likely to be random). Furthermore, the prevalence rates for IHD and stroke are given with respect to people of all ages registered with the GP, and Diabetes for people aged 17+, which means that we may be under-estimating the noise attributable burden of disease for adults (20 +) in England as the underlying disease prevalence rates include a population of children and adolescents who are at lower risk. Furthermore, we may be under-estimating the most in London as it has the lowest regional median age in England (Office for National Statistics, 2020b).

As discussed in Section 2.3, we applied population noise exposure distributions from strategic mapping undertaken in 2012 to the population distribution (i.e., population at risk) within LAD in 2018. We thus, assumed that the spatial distribution and magnitude of the noise exposure was stable between these time points. Furthermore, while the availability of strategic noise mapping data greatly facilitates burden of disease assessments at national, regional and local levels, the downside is that only major sources are captured and the lower exposure threshold of 50 dB L_{den} and L_{night} mean that we are likely under-estimating the total noise attributable burden of disease in England (as we showed in our Sensitivity Analysis in Section 3.5). Lastly, while people spend the majority of their time indoors, we were unable to account for geographical differences in the ingress of outdoor noise into indoor environments due to variations in building characteristics and ventilation practices across the country.

5. Conclusions

We produced the first geographical assessment of the burden of disease due to noise for England. Our study shows that transportation noise exposure, particularly from road-traffic sources, is responsible for a significant disease burden in England and this varies unequally across regions and LADs. Our work provides useful nationwide information for identifying areas with varying burdens of disease due to different noise sources and for setting priorities in environmental health research, policies, and interventions. Quantifying the disease burden is the first step in building an environmental, social and economic case for action that takes into account the views and expertise of national and local government, environmental health practitioners, acoustics specialists, industry, the third sector and affected citizens. The epidemiological evidence on noise and health continues to develop, and disease burden estimates are likely to change as more data from good quality longitudinal studies becomes available. Lastly, we showed that exposure data derived from strategic noise mapping models can lead to underestimates in the attributable burden of disease if not all transport sources are accounted for and if lower levels of noise exposure are omitted.

CRediT authorship contribution statement

Calvin Jephcote: Data curation, Formal analysis, Methodology, Software, Writing – original draft, Writing – review & editing. **Sierra N. Clark:** Data curation, Formal analysis, Visualization, Methodology, Writing – original draft, Writing – review & editing. **Anna L. Hansell:** Methodology, Investigation, Writing – review & editing. **Nigel Jones:** Conceptualization, Data curation, Project administration, Supervision, Writing – original draft, Writing – review & editing. **Yingxin Chen:** Formal analysis, Writing – review & editing. **Claire Blackmore:** Formal analysis, Writing – review & editing. **Katie Eminson:** Formal analysis, Writing – review & editing. **Megan Evans:** Formal analysis, Writing – review & editing. **Xiangpu Gong:** Formal analysis, Writing – review & editing. **Kathryn Adams:** Data curation, Writing – review & editing. **Georgia Rodgers:** Writing – review & editing. **Benjamin Fenech:** Conceptualization, Investigation, Methodology, Project administration, Resources, Supervision, Writing – original draft, Writing – review & editing. **John Gulliver:** Conceptualization, Investigation, Methodology, Project administration, Software, Supervision, Visualization, Writing – original draft, Writing – review & editing.

Declaration of Competing Interest

The authors declare that they have no known competing financial interests or personal relationships that could have appeared to influence the work reported in this paper.

Data availability

The epidemiological, population, and health data are all in the public domain and cited throughout the paper. The ENR strategic noise exposure data is available in the public domain aggregated in 5 dB bands with lower thresholds of 55 dB L_{den} and 50 dB L_{night}. .

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Appendix A. Supplementary material

Supplementary data to this article can be found online at https://doi.org/10.1016/j.envint.2023.107966.

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