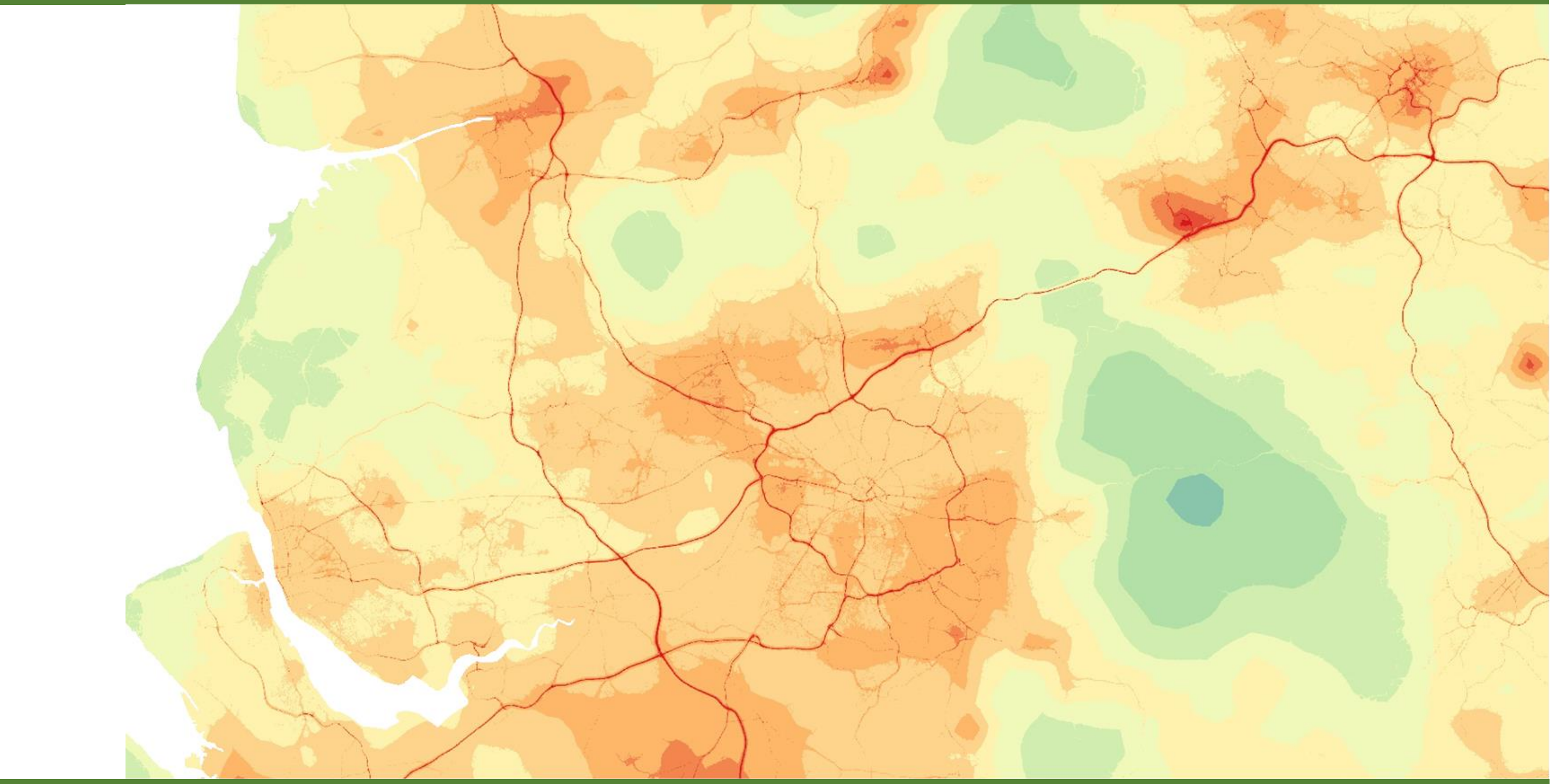
AIR QUALITY EXPERT GROUP

# Differentials in air pollutant exposure across communities and regions in the UK



Prepared for:

Department for Environment, Food and Rural Affairs;
Scottish Government; Welsh Government;
and Department of Agriculture, Environment and Rural Affairs in Northern Ireland

This is a report from the Air Quality Expert Group to the Department for Environment, Food and Rural Affairs; Scottish Government; Welsh Government; and Department of Agriculture, Environment and Rural Affairs in Northern Ireland, on the current state of scientific and technical knowledge of the differentials that exist in air pollution emissions and atmospheric concentrations across the United Kingdom and related issues of relevance to air quality management. The information contained within this report represents a review of the understanding and evidence available at the time of writing.

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Freephone Air Pollution Information Service 0800 556677

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## Terms of Reference

The Air Quality Expert Group (AQEG) is an expert committee of the Department for Environment, Food and Rural Affairs (Defra) and considers current knowledge on air pollution and provides advice on such things as the levels, sources and characteristics of air pollutants in the UK. AQEG reports to Defra's Chief Scientific Adviser, Defra Ministers, Scottish Ministers, the Welsh Government and the Department of Agriculture, Environment and Rural Affairs in Northern Ireland (the Government and devolved administrations). Members of the Group are drawn from those with a proven track record in the fields of air pollution research and practice.

AQEG's functions are to:

1. Provide advice to, and work collaboratively with, officials and key office holders in Defra and the devolved administrations, other delivery partners and public bodies, and EU and international technical expert groups;
2. Report to Defra's Chief Scientific Adviser (CSA): Chairs of expert committees will meet annually with the CSA, and will provide an annual summary of the work of the Committee to the Science Advisory Council (SAC) for Defra's Annual Report. In exception, matters can be escalated to Ministers;
3. Support the CSA as appropriate during emergencies;
4. Contribute to developing the air quality evidence base by analysing, interpreting and synthesising evidence;
5. Provide judgements on the quality and relevance of the evidence base;
6. Suggest priority areas for future work, and advise on Defra's implementation of the air quality evidence plan (or equivalent);
7. Give advice on current and future levels, trends, sources and characteristics of air pollutants in the UK;
8. Provide independent advice and operate in line with the Government's Principles for Scientific Advice and the Code of Practice for Scientific Advisory Committees (CoPSAC).

Expert Committee Members are independent appointments made through open competition, in line with the Office of the Commissioner for Public Appointments (OCPA) guidelines on best practice for making public appointments. Members are expected to act in accord with the principles of public life.

Further information on AQEG can be found on the Group's website at: https://www.gov.uk/government/policy-advisory-groups/air-quality-expert-group

and https://uk-air.defra.gov.uk/research/aqeg/

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National Centre for Atmospheric Science, University of York

### Members

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National Centre for Atmospheric Science, University of Manchester

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UWE Bristol

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Imperial College London

**Professor David Carslaw**

Ricardo Energy and Environment and University of York

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Aether Ltd

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Imperial College London

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**Professor Mathew Heal**

University of Edinburgh

**Dr Ben Marner**

Air Quality Consultants Ltd

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University of Sheffield

**Dr Eiko Nemitz**

UK Centre for Ecology & Hydrology

## *Ad hoc* members

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National Centre for Atmospheric Science, University of York

**Professor Anil Namdeo**

Northumbria University

**Professor David Topping**

University of Manchester

**Dr Daniela Fecht**

Imperial College London

## *Ex officio* members

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National Atmospheric Emissions Inventory: **Dr Tim Murrells** (until 2023)**, Rob Stewart** (2024-), Ricardo Energy and Environment

Heavy metals monitoring network: **Dr Nick Martin** (2022-2024), **Dr Andrew Brown** (2024-), National Physical Laboratory

Quality Assurance and Quality Control of the automatic urban network and the non-automatic monitoring networks: **Dr Paul Willis**, Ricardo Energy and Environment

Environment Agency: **Dr Rob Kinnersley**

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Welsh Government

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**Michelle Brailey-Balster**

Department for Environment, Food and Rural Affairs

**Jenny Hudson-Bell**

National Centre for Atmospheric Science / University of York

# Differentials in UK air pollution emissions and concentrations

## Contents

# Executive summary

This report considers the differentials in air pollutant emissions, concentrations and exposure across the geographies and communities of the UK. It lays out the extent of evidence on these differentials and highlights gaps in our knowledge.

A complete understanding of the possible causes and impacts will only be achieved through a multidisciplinary synthesis of the evidence base that is beyond the remit of AQEG. Our report synthesises aspects of this complex socio-environmental and health challenge.

We hope that this report will be a catalyst for future research to better understand differentials in relation to UK air quality and identify opportunities to maximise the benefits from air pollution interventions.

## Spatial differences in air pollution across the UK

Substantial differences exist in the concentrations of outdoor air pollutants across the United Kingdom and in the distribution of anthropogenic and biogenic emission sources.

Gradients in outdoor air pollutants concentrations exist in part due to the different effects of weather experienced in different UK regions and the proximity to regional and transboundary sources. For most air pollutants the northern and western parts of the UK experience lower concentrations than the south and east, although there is considerable variation within regions, cities and even within areas that share the same postcode scale. Variability arises due to proximity to individual emission sources, population density, wetter and windier weather in the north and west and differences in natural emissions and background concentrations, for example the extent of the influence of cleaner Atlantic oceanic air. These national and regional differentials in air quality differ depending on pollutant with nitrogen dioxide ($NO_2$) varying substantially over short distance scales (as small as tens of metres or less), whilst $PM_{2.5}$ and $O_3$ are more broadly distributed with some local variability superimposed on an elevated regional concentration. On average $NO_2$ and $PM_{2.5}$ are highest for urban populations, with $O_3$ at its greatest in the countryside.

The spatial resolution used in analyses can significantly affect conclusions on the linkages observed between socioeconomic status, demography and ethnicity and air pollution exposure. Studies that use coarse spatial averaging can suggest opposite patterns to those which use more granular data.

## Social determinants and environmental justice

Beyond transboundary and weather factors that can influence the distribution of air pollutants many international studies have shown that low income and deprived communities often live in areas with higher air pollution concentrations and higher local emissions.

Much of the available research literature relates to North America where this is framed as an indicator of what has been termed environmental injustice. There is a more limited research literature of direct relevance to the UK; here air quality disparities associated with social deprivation are not universal and can differ by pollutant. Whilst in North America, research demonstrates inequity of exposure to air pollution with the greatest burden falling on the most vulnerable, evidence in UK and European studies is more mixed, sometimes showing weaker relationships between air pollution concentrations, exposure and socioeconomic factors. In a policy context, the concept of environmental justice is widely used to assess differential effects of interventions and policy in North America, but it is not widely researched or applied in the UK.

In some UK studies there is evidence that socioeconomic disparities in exposure to outdoor $NO_2$ and $PM_{2.5}$ arise from multiple factors and are predominantly an urban phenomenon. There is some evidence of higher air pollution concentrations being experienced by certain minority ethnicity groups and some age groups.

An important issue linked to environmental justice is that whilst more deprived communities in the UK have been estimated to be exposed to greater ambient air pollution arising from road transport, those communities make a smaller contribution to transport emissions than less deprived groups.

Most policy interventions for improving air quality focus on reducing primary emissions. In the UK, local emissions are on average greater for more deprived communities for both $NO_x$ (the collective term for NO and $NO_2$) and $PM_{2.5}$, arising from sources including road transport, space heating, construction and industry. Local emissions of $NO_x$ and $PM_{2.5}$ are greater in geographic locations with a high fraction of residents from minoritised ethnicity groups, when compared to locations with majority white populations and with matched socioeconomic status.

## Travel, work and study

There is some UK-based evidence that when individuals’ places of work or study are included in the estimation of exposure this reduces some of the disparities that are observed on average across demographics such as age and deprivation. This can be rationalised as resulting from more affluent individuals with homes in lower pollution locations spending time each day studying or working in more polluted urban centres.

Travel and place of work or study are therefore significant factors that can influence overall exposure to air pollution. Using motorised transport modes generally leads to the greatest exposure with in-vehicle cabin concentrations of air pollutants often greater than those experienced by active travellers. For those cities with underground / subway systems, this can be the travel mode with highest $PM_{10}$ concentrations. Lowest exposure for rail and active travellers has been reported where railway lines, cycle lanes, and footpaths were located away from roads. Greater use of buses by the least well off and longer journey times by bus can lead to elevated exposure. AQEG note that choice of travel mode, place of work or study and occupation are themselves influenced by a combination of geographic, demographic and socioeconomic factors.

Occupational exposure to pollution (i.e. direct exposure to pollutants in work microenvironments such as mining, construction etc) is outside of the AQEG terms of reference and is governed by Health and Safety regulation. We note however that certain occupations can experience elevated exposure to ambient air pollution by virtue of the place of work being outdoors in urban and roadside environments, for example for professional drivers and construction workers.

## Housing quality

There are some well-established associations between lower quality of housing, higher social deprivation and increased exposure to indoor air pollution, particularly to bioaerosols arising from damp and mould in homes.

Increased exposure indoors is also associated with personal activities such as cooking and smoking and can be amplified by a higher occupant density. These emissions may lead to increased indoor concentrations following measures taken to improve energy efficiency which can sometimes reduce building ventilation and the exchange of indoor air with air from outside.

If trends towards improved energy efficiency in the UK housing stock also reduces ventilation this has the potential to lead to an increase in indoor air pollution exposure. AQEG note that high deprivation households often have limited agency in mitigating exposure to indoor exposure.

More affluent households may have on average higher domestic chemical consumption and are also more likely to use stoves burning wood as a non-primary heating source.

Evaluating associations between indoor air pollution and other social or demographic factors is however limited by a lack of UK measurement data, meaning that proxies such as ventilation rates, energy efficiency or reports of damp are often the only available related metrics. Without good underpinning measurement data it is difficult to develop and test the effectiveness of potential policy or technical interventions.

## Existing air pollution policies and regulations

A substantial range of different regulations, strategies and policies exist with explicit aims to improve air quality in the UK. These include regulations that limit emissions at source from products and installations and decision-making processes that determine where new sources of pollution may be allowed.

Beyond direct interventions that aim to reduce emissions or exposure, national and local government planning and development decision-making also considers impacts on air quality. Consideration of air quality when planning new development generally focuses on air quality objectives and limit values.

There is limited evidence for the extent to which local authority and city-level air quality management actions have exacerbated or reduced the disparities that exist between different regions and communities. However, there is now explicit guidance available for Local Air Quality Management to consider disparities resulting from differential exposures to air pollution, as well as the implications of policy implementation for those with least ability to limit their exposure or change their behaviours. This includes working across policy disciplines and implementing inclusive communication strategies. Evaluation of recent national-level policy and regulation (for example in the Environment Act, 2021) has considered disparity impacts as a function of the Indices of Multiple Deprivation (IMD).

## UK datasets

The UK has many comprehensive open-access government datasets including highly detailed annually updated air pollution emissions inventories, extensive outdoor air pollution monitoring networks and detailed decadal population census information (most recently from 2021). This provides opportunities for the identification of associations that might not be possible in other countries (for example disparities by detailed ethnicity), and the disaggregation of possible confounding effects.

# Recommendations

## Building upon existing UK datasets for better outcomes

1. **Assessment of changes in community and regional differentials in ambient air pollution should be undertaken when considering new policy options**. Most policy interventions for improving air quality focus on reducing primary emissions. Substantial differentials exist in the distribution of current emissions across the UK; on average, communities with higher levels of deprivation live in areas with higher localised emissions of $PM_{2.5}$ and $NO_x$. The effect that a change in local emissions might have on reducing or increasing emissions differentials can be relatively straightforwardly evaluated drawing on datasets from the National Atmospheric Emissions Inventory (NAEI) and Office for National Statistics (ONS). However, emissions are often not a good proxy for local ambient concentrations and exposure, and policies may have complex consequences in other domains.

2. **Further research is required to better understand the current disparities in air pollution exposure experienced by different communities and how this varies between different towns, cities and regions**. Whilst there is a substantial international literature on differences in air pollution concentrations experienced by people according to their socioeconomic status, ethnicity or health vulnerability, there is only limited information directly relevant to the UK. Where studies have been undertaken in a UK context they sometimes apply only to parts of the country (e.g. England and Wales), to specific cities, or are now somewhat outdated.

3. **Improved tools, methods and datasets are needed to support evaluation of potential disparities in air pollution exposure between different communities.** Improved tools are needed to more fully account for other pre-existing gradients in concentrations arising from factors such as weather and population distribution, for example through the integration of regional scale models with local observations and local demographic and socioeconomic data. This is particularly relevant to longer lived pollutants such as $PM_{2.5}$ and $O_3$ for which large geographic differentials in concentrations exist due to meteorological effects (e.g. higher rainfall and windspeeds) and where proximity to transboundary sources (e.g. distance to mainland Europe) is a controlling factor.

4. **New resources should be developed to assess the intersectionality between air pollution and other environmental factors and to evaluate impacts on specific communities, geographies and demographic groups.** There are multiple opportunities for new data and statistical tools to support the optimisation of air quality interventions should reduction in air pollution differentials be a policy objective. These assessments could encompass ambient noise as well as socioeconomic factors and metrics of health vulnerability including individual and cumulative impacts.

5. **There are opportunities to use data on traffic flows, mobility, health vulnerabilities, employment and education to gain greater insight into which communities are**

**most impacted by air pollution, and how actions may be targeted to ameliorate those effects.** Whilst the UK has rich public sector data resources not all of these have been exploited fully for research or to inform air quality policymaking. Machine learning provides new opportunities to use such diverse data and should provide a means for government to assess air quality interventions more routinely, and other policy outcomes, which may have differential effects by region, communities or demography.

6. **Future UKRI and Defra research programmes should be encouraged to produce datasets and tools to provide actionable evidence to reduce current and future differentials.** This is likely to involve engaging across multiple academic and professional disciplines and communities, considering air pollution as a contributor to a wider discussion on drivers of, and actions to address, heath disparities in the UK.

## Travel, work and study

7. **AQEG recommend that impact assessments incorporate air pollution exposure during travel alongside socioeconomic metrics**. Substantial air pollution exposure occurs during the use of transport, including exposure during walking or cycling. This would enable the identification of communities or vulnerable people that experience the greatest air pollution exposure during travel. By including emissions, impact assessments could also evaluate issues relating to environmental justice, that is the balance between who pollutes and who experiences pollution.

8. **AQEG recommend the development of an improved evidence-base on air pollution differentials by occupation.** Evidence of how occupation impacts on exposure to ambient outdoor pollution is rather limited in a UK context, although clearly some groups such as professional drivers and construction workers are likely more exposed due to the nature of their working environment. There are likely to be occupational differentials in air pollution exposure also arising from different indoor workplaces, currently managed via occupational health guidance and health and safety at work regulation. Since outdoor air impacts on indoor air quality at work, increased coordination of research, evidence and policy response across government departments to improve indoor air quality in the workplace should be explored.

## Future delivery of improved air quality

9. **AQEG recommend the evaluation of differential effects of policy and actions to reduce emissions in key sectors beyond road transport.** Emissions from road transport remain an important determinant of urban air quality, and currently give rise to substantial differentials in exposure to ambient $NO_2$ between communities. Air quality interventions directed at urban road transport will continue to have a role to play in reducing differentials in pollution experienced by different regions and groups for many years to come. However, as exhaust emissions from vehicles decline other sources such as vehicle non-exhaust emissions, building heating, airports, railways, construction and industrial sources will become increasingly important in determining who is exposed to air pollution.

10. **New development that results in changes to ambient concentrations should consider the differentials in air quality experienced by different communities.** Current local government systems for planning and development consider air quality as a potential effect, however this is generally viewed as consequential to decision-making only if regulatory limit values are at risk of being exceeded. This may be particularly relevant in decision-making around the installation of new point sources such as energy-from-waste plants.

## Housing quality and the indoor environment

11. **New data and monitoring programmes are needed to evaluate air pollution exposure in typical UK indoor environments and how these vary according to socioeconomic and other factors.** The effects of housing quality on health are well documented. A substantial fraction of daily air pollution exposure takes place indoors including to classical pollutants such as $NO_2$, VOCs and $PM_{2.5}$ and to moulds and spores arising from damp. However, the extent to which there are differentials linked to geographic factors, weather, socioeconomic status, ethnicity or demographics is highly uncertain due to limited measurement data. It is unlikely that evidence from other countries will be applicable to UK given the unique nature of the national building stock and prevailing climate.

12. **Further basic research is needed to support a more complete process-based knowledge of the links between variables such as indoor activity, housing type and ventilation with indoor air pollution.** This is an essential pre-requisite to support the predictive models that can then simulate indoor exposure and the effects of new products, abatement technologies, policy and regulation.

13. **There is a need for systematic evaluation of air quality in different building types in the UK. This should include the impacts of regulations and guidance for the construction of energy-efficient buildings and retrofits.** Evaluating the potential differential impacts on changing indoor air quality driven by decarbonisation of homes is a particularly important and urgent evidence gap. Whilst some decarbonisation actions such as electrification (e.g. replacing gas boilers with heat pumps) remove key pollution sources from homes and communities, the effects of energy efficiency measures and ventilation changes on damp and mould, and the accumulation of indoor emissions of primary pollutants such as $NO_2$, $PM_{2.5}$ and VOCs is uncertain, particularly for retrofits.

# Chapter 1 – Introduction

The distribution of air pollutants across the UK is very uneven with some locations experiencing higher concentrations in ambient air than others. The degree of heterogeneity in concentrations depends on the pollutant characteristics, the distribution of emissions (and by extension transport systems, industry, homes and so on) and external factors such as geography and weather. Shorter-lived air pollutants such as nitrogen oxides ($NO_x$) can show steep spatial gradients in concentration, highest close to sources such as roads and combustion, but that are much lower even just a few hundred metres away in the urban background. Longer-lived and secondary pollutants such as particulate matter ($PM_{2.5}$) and ozone can accumulate in the air over several days and can be found more evenly distributed across the country as a whole. A consequence of an uneven distribution of air pollution is an uneven distribution of health and ecosystem impacts with some locations and communities experiencing higher pollution in their localities than others.

Disparities in exposure to air pollution, a combination of where someone lives, along with where they may spend time during the day, including commuting, at school or work for example, arise because of a complex set of phenomena that are explored further in this report. The topic is inevitably a sensitive one since for one individual to be exposed to higher pollution than another immediately then raises questions about fairness and an uneven or unjust differential in burden of harm.  Whilst some differences in air pollution do indeed arise due to structural differences in where people and pollution are located in the UK, it is not the only factor at play. In this report, AQEG surveys recent data and literature that provides insight into the various drivers of differentials in concentrations and emissions, both for outdoors and indoors, and as a function of where people live, how they travel and for certain occupations who work outdoors.

A rich variety of data sources and methodological approaches can support analysis on disparities in air pollution including long-term measurements made in the Defra, Devolved Administrations and Local Authority monitoring networks, research-led satellite, mobile and sensor measurements, and highly detailed information on emissions from the National Atmospheric Emissions Inventory. These traditional air pollution data resources can be supplemented by additional complementary information on UK geographic and land-use characteristics, population information (for example from the UK census), and other administrative, demographic and socioeconomic data. In combination these datasets underpin most research literature reporting on the factors that drive the heterogeneity in atmospheric concentrations.

In very broad terms some of those controlling factors arise from geographical characteristics and others are associated with population, economic activity and choices (planned and unplanned) made around where polluting sources are located. Chapter 2 focuses on summarising data and methods that have been applied previously, drawing on international studies and insight from those that might be applied in a UK context. The body of peer-reviewed work that specifically is concerned with the UK is limited and not all conclusions from studies based on international datasets and experiences can be directly translated into a UK setting. Positively however, the UK has some very comprehensive open-access Government datasets including highly detailed census information, which provide

opportunities for disaggregation of causes and effects that might not be possible in other countries.
A significant driver of differentials in ambient air pollution experienced in the UK are the effects of weather and geography. The broad spatial differences in current air pollution concentrations are examined in Chapter 3 drawing on measurements and modelling. For example, there are pronounced effects of experiencing, on average, wetter and windier conditions in the north west compared to the drier south east of the country. Proximity to the clean Atlantic Ocean is a major factor in determining the $PM_{2.5}$, $NO_2$ or $SO_2$ that someone might experience in the UK. On average, cleaner air flows into locations in the west of Northern Ireland and west coast of Scotland. In contrast, the south east of England is closer to mainland European emissions and a transboundary flow of pollution can have a substantial influence in raising concentrations broadly in this region. Population density, which can be a reasonable proxy for the intensity of certain activities and emissions of primary pollutants (e.g. $NO_x$, VOC, PM) is also highly variable in the UK, from a densely populated megacity in London, to sparsely populated regions of northern England, Wales and Scotland. Here the lifetimes of pollutants become important, with highest $NO_x$ concentrations being closely linked in geographic terms to sources whilst $PM_{2.5}$ and ozone are more evenly distributed across the country.

The national atmospheric emissions inventory (NAEI) provides a particularly detailed understanding of how the emissions of air pollutants are currently distributed across the country. Whilst ambient concentrations are what cause harms to people, the location of emissions provides insight into structural aspects of air pollution, a key factor since ultimately local and national air pollution policies can largely only influence where emissions occur, and how much. Where emissions occur, and why, is complex but they do to a degree reflect long-term national policy choices. Sometimes emissions follow the movement of people, for example growing urban populations may lead to higher transport and space heating emissions. The location of a major industrial source of pollution may attract people for employment. In Chapter 4 the distribution of emissions sources in the UK is examined, together with their links to socioeconomic factors and demographics. It is here that some important issues around environmental justice and structural unfairness become visible. In broad terms those populations in the UK experiencing higher levels of deprivation tend to live closer to higher levels of primary emissions. Those same communities are often responsible for only a small fraction of the air pollution they experience.

Emissions provide insight into where primary sources are located relative to people, it is ambient concentrations that drive exposure and ultimately harm. The distributions of concentrations in this context are examined in Chapter 5. At the global scale there is frequent reporting of higher ambient concentrations of $PM_{2.5}$ and $NO_2$ occurring close to those experiencing higher levels of deprivation and to vulnerable and/or minoritized groups. The peer-reviewed literature that is available specifically examining effects in the UK is more limited and the picture is nuanced, with disparity effects most pronounced in cities for $NO_2$ and $PM_{2.5}$. The demographic make-up of some cities in the UK, for example with substantial transient younger populations, and higher income commuting daytime populations, to a degree attenuates differences between different socioeconomic groups, although some inequities in exposure specifically to road transport emissions have been reported in literature. It is important however to be aware that comparison of international studies between countries, and to a degree national studies, is further complicated because many different ways can be used to characterize socioeconomic deprivation.

Whilst there is a substantial global literature examining the distribution of outdoor air pollution across different populations, the evaluation of differences and disparities in exposure in indoor environments is somewhat more limited, particularly in high income countries. Here there are several potentially competing factors at play, and the effects and distribution of pollution may well be specific to individual pollutants. There are well-understood links between higher levels of deprivation and poor housing quality which in turn can lead to exposure to damp and higher concentrations of biological particles, such as moulds and spores. Poor quality homes may have poorly maintained heating systems and give rise to elevated pollutants such as CO and PM. However, for some other air pollutants, for example VOCs, indoor concentrations may be elevated in higher income households, a function of greater domestic chemical consumption and better insulated houses with lower rates of air exchange. In Chapter 6 both chemical and biological air pollutants indoors are considered alongside reported recent statistics on the UK housing stock and studies on pollution concentrations of relevance to UK settings.

Whilst travelling makes up only a modest fraction of most people's day it can be responsible for a substantial fraction of exposure to air pollution. Differences in air pollution experienced have been extensively studied as a function of travel mode, for example car, rail, bus, active travel. There are however socioeconomic and demographic dimensions to exposure to air pollution during travel since different groups are more or less likely to use particular modes of transport. Although this is a relatively underexplored area, Chapter 7 identifies relevant studies and datasets that provide some insight in these factors. A key issue however in evaluating the evidence on impacts of travel and exposure to air pollution is that there have been profound changes in vehicle emissions in recent years. Studies that relate to road transport fleets from 10 or more years ago may give a very different picture than if those same analyses were recreated today. Whilst occupational exposure to pollution is a topic in its own right, and outside of the AQEG remit, many jobs have the outdoor ambient environment as the workplace. In these cases broader trends and distribution of ambient air pollution define to a degree occupational exposure. Occupations particularly impacted in this regard include professional drivers and construction workers.

The improvement in air quality over past decades has been a result of combined policy and technological efforts to reduce emissions. Inevitably policies that tackle emissions at source and by sectors will have unequal impacts on ambient air since those sources will be unevenly distributed. There is considerable emphasis now being placed on considering how policies may account for disparity in effects and address historical structural inequality in environmental benefits and harms. This forms a core objective within the Defra 25 Year Environment Plan. Within air quality, three broad policy approaches are taken to improve conditions, with interactions between them; i) reducing emissions from specific sources, ii) ensuring that ambient concentrations meet prescribed legal objectives for air quality and iii) reducing concentrations across the population as a whole. Each brings benefits but the scale of effects on different groups will be heterogenous. One particularly impactful area of intervention is the extent to which land-use planning and development decisions consider air quality and the requirement to consider how air quality might be impacted by development, who would be impacted, and by how much. Chapter 8 sets out the growing body of national objectives, and potential practices, for evaluating air pollution changes arising as a result of policy choices.

# Chapter 2 – Data sources and techniques

## 2.1. Data Sources

Before reviewing the differentials in air pollution that exist across the UK, it is first important to consider what data sources are available, which of those datasets have been used in previous academic and government studies, and what limitations or gaps exist. Datasets that inform on this topic go beyond those used typically by AQEG, including socioeconomic and demographic data, as well as spatial and geographic information.

### 2.1.1 Air quality datasets

Differentials in the emissions and ambient concentrations of air pollution exist for many different types of air pollution. Commonly studied pollutants include both the emissions and concentrations of particulate matter (PM), nitrogen dioxide ($NO_2$), ozone ($O_3$), sulphur dioxide ($SO_2$), carbon monoxide (CO), volatile organic compounds (VOCs), black carbon (BC) and hazardous air pollutants (HAPs), such as asbestos, methylene chloride, and heavy metals. Datasets on air pollution can include measurements made for regulatory and compliance purposes, for example the Automatic Urban and Rural network (AURN), research community datasets on concentrations or models of either emissions, concentrations or both. Supporting datasets can include meteorological data, either measured at monitoring sites, drawn from weather models or data reanalyses.

Sources of data reported in literature are highly varied and a lack of data at a particular spatial resolution frequently becomes a limiting factor for some methodologies. In the UK, studies of air pollution differentials have often used government modelled emissions or concentrations of pollutants alongside national Census variables, based on census geographies, such as Lower-level Super Output Areas (LSOAs). Modelled air quality datasets can be produced over a range of resolutions, from 10s of kilometre gridboxes for hemispheric modelling of long-lived pollutants down to a few metres resolution using dispersion models of street canyons. Government air quality and emissions datasets are typically produced at 1 km x 1 km spatial resolutions requiring manipulation of the air quality data into (population-weighted) averages that align with the census geographies. Neither modelled air quality nor census geographical units align with the local authority areas that have responsibilities for local management and there can be significant differences in terms of concentration/emissions values within these spatial units.

### 2.1.2 Socioeconomic datasets

In a systematic review of studies across Western Europe, ecological (or areal) studies (see section 2.2) and studies using individual data both show high exposure to poor air quality is linked to deprivation and lower economic position, but that ecological studies often demonstrate a ‘U-shaped’ curve with most and least deprived areas having higher levels of air pollution (Fairburn *et al*. 2019). Fairburn et al. identified that a lack of small area statistics with good socioeconomic, socio-demographic or index data in some EU countries restricted the use of ecological studies more widely and that the age of the data used in some studies might not reflect present-day conditions.

In the UK, separate Indices for Multiple Deprivation are available for England, Scotland, Wales and Northern Ireland (Index of Multiple Deprivation (IMD) | CDRC Data), which use weighted combinations of variables within domains relating to income; employment; health deprivation and disability; education, skills training; crime; barriers to housing and services; and living environment. Some UK studies of air pollution and deprivation have used the income domain of Multiple Deprivation Indices (e.g. Brunt et al., 2017; Fecht et al., 2015), to avoid spatial autocorrelation with health and environment domains.

Other proxies for socioeconomic status used in air pollution differentials studies include estimated median household income (e.g. Experian) and poverty indices (e.g. Breadline Britain Index, Carstairs Index, and Townsend Index), economic activity, or vehicle ownership/access (Mitchell and Dorling, 2003; Wheeler, 2004; Barnes et al., 2019). The Office for National Statistics present data on socioeconomic disparities across the UK in a two-part interactive series: (Mapping inequality in the UK (ons.gov.uk)).

### 2.1.3 Demographic datasets – ethnicity and age

Substantial differentials in air pollution have been reported in global research literature that are linked to datasets of ethnicity (defined, for example as minority ethnicity groups, immigrants or foreign-born mothers). Minoritised ethnicity groups are frequently associated with having exposure to higher concentrations of air pollution, however disaggregation of ethnicity from deprivation is not always possible (Fairburn et al., 2019) since information is not always collected concurrently. Differentials in air pollution exposure derived from datasets related to education level, occupation and gender have shown mixed results (Fairburn et al., 2019).

As an example, Fecht et al*.,* 2015 used data from the 2001 UK Census to determine that neighbourhoods in England with more than 20% non-White residents had higher mean $PM_{10}$ and $NO_2$ after adjustment for deprivation and demographics. In London, Census data from 2011 alongside IMD data for London was used to establish that $NO_2$ concentrations were 16-27% higher in areas with higher ethnic minorities populations and that 31-35% of areas with the highest proportion of black and mixed/multiple ethnicities were in areas with higher levels of air pollution, cf. 15-18% for Asian ethnic groups and just 4-5% for white ethnic groups (Williamson et al., 2021).

Linking air pollution datasets with demographic data can also highlight differentials in relation to air pollution exposure. For example, age differentials were reported by Barnes et al. (2019) and Mitchell and Dorling (2003), based on analysis of census data, reporting that areas with greater numbers of under-fives and adults aged 20-44 were associated with elevated $NO_2$ concentrations, whereas areas with more over-45s tend to have better air quality. $NO_2$ concentrations in areas with more young adults were estimated to be twice the average, and with road $NO_x$ emissions five times greater (Barnes et al., 2019).

It is worth noting that many of these studies rely heavily on census data which is only collected every 10 years and therefore do not necessarily reflect changes in population characteristics aligned to changes in air pollution in intervening periods.

## 2.2. Techniques and approaches

An array of different methods has been employed to estimate air pollution emissions and concentrations (as illustrated in the following studies) with the aim of uncovering links between exposure differentials, disparities and sociodemographic vulnerability. These methods broadly fall into two categories: quantitative and qualitative approaches.

### 2.2.1 Quantitative approaches

Quantitative approaches use numerical data to establish the link between air pollution and health-related social parameters such as income, race/ethnicity, and educational attainment. These methods have been commonly categorized into proxy, monitor-based, statistical, process-based, and ecological and area-based methods (e.g. Gardner-Frolick et al., 2022, Casey et al., 2023). Each method is mostly differentiated by the techniques applied to estimate air pollution concentrations and their associations with sociodemographic factors.

**PROXY METHODS**

Proxy methods are among the most well-established and most frequently used in environmental justice studies and offer an approximation of air pollution hazards by focusing on factors upstream of pollutant concentration and exposure in the causal impact chain. Studies using these methods are primarily centred on the location of emissions, and they are based on ambient concentrations, particularly at the place of residence, as a proxy for exposure. The methods use source location or emissions to formulate a proxy estimate for concentration.

- *Unit-Hazard Coincidence*: This method identifies and compares the number of hazards within a defined geographic boundary, offering a straightforward means to assess communities based on sociodemographic characteristics. It originated from the first major US environmental justice study on air pollution in 1980s (United Church of Christ, 1987) and remains widely employed to capture the adverse effects of proximity to environmental hazards, particularly HAPs from industrial sources (e.g., Chakraborty et al., 2014). Its simplicity facilitates practical application, making it valuable when monitoring data or validated modelling frameworks are lacking. However, limitations include potential spatial obscuration of hazard-population relationships, a focus on primary pollutants, and variability between studies.

- *Proximity Analysis*: This method estimates pollution exposure by overlaying a buffer of a certain distance around a hazard's location, defining "exposed" and "unexposed" areas. It has widely been used to study the impact of toxic releases of HAPs and industrial pollutants, traffic and infrastructure emissions (e.g., Perera et al., 2013; Gaffron et al., 2015; Boda et al., 2023). While advantageous for capturing the broader impacts of living near environmental hazards, this method does not directly define concentrations or exposure. Compared to the Unit-Hazard Coincidence method, it may offer more accurate influence area delineation but requires user-defined geographic boundaries, potentially increasing complexity in sociodemographic data adjustment.

## MONITOR-BASED METHODS

Monitor-based methods in air pollution research leverage diverse monitoring instruments, from stationary monitors, mobile units, wearable sensors, low-cost sensors, and satellite observations (e.g. Ottinger, 2010, Tonne et al., 2018, Kerr et al., 2023). These methods have been traditionally applied by local and national authorities and research institutions. However, the widespread use of low-cost sensors (both real-time and diffusion samplers) have increased the involvement of communities, offering more access to air quality assessment as well as providing alternative means for personal exposure estimates (Ottinger, 2010). Real-time lower-cost sensors are however typically less accurate than monitors used for regulatory purposes, and there is ongoing evaluation of their effectiveness. Monitor-based data are typically analysed using techniques that include exposure and interpolation.

- *Personal Exposure*: This method aims to capture the unique and individualized exposure to air pollution a person faces in their daily life and has been applied to a wide range of pollutants, from PM to HAPs (Gardner-Frolick et al., 2022; Casey et al., 2023). Using wearable monitors or placing stationary monitors where individuals spend the most time, such as at home or work, personal exposure studies examine the specifics of what people breathe. While offering a highly accurate portrayal of an individual's actual exposure over a short period of time, challenges arise due to resource constraints, limited pollutant coverage of wearable monitors, and the difficulty of obtaining comprehensive datasets on a community scale. Despite these challenges, personal exposure studies, often used in health-focused research (e.g. Brody et al., 2009), can integrate with community-based initiatives on air quality.

- *Interpolation*: Spatial interpolation methods rely solely on monitoring data, employing arithmetic processing to estimate pollutant concentrations at unsampled locations. They are most applied to criteria pollutants such as PM (including BC), $O_3$, and $NO_2$, primarily due to the comprehensive regulatory monitoring network and their significant health implications (e.g. Gaffron et al., 2015, Moreno-Jiménez et al., 2016). Whilst simple and capable of capturing similar features as more complex methods, interpolation faces challenges in environmental justice research due to spatial gaps in monitoring networks and the inherent spatial smoothing introduced by interpolation. While resource-efficient, the effectiveness of interpolation depends on data availability and may overlook localized areas of elevated pollution, impacting its ease of use.

- *Satellite Remote Sensing*: Satellite measurements are a relatively recent technique for assessing surface air pollution. Using optical depth and spectral measurements, these sensors can estimate pollutant concentrations, although this process is complex due to its computational intensity and the variety of methods available (Sayyed et al., 2024). This approach allows estimation of concentrations of a wide range of pollutants including PM, $O_3$, $CH_4$, CO, $SO_2$, and $NO_2$. Satellite data have been applied to detect industrial flaring (e.g. Johnston et al., 2020), estimate long-term concentrations over large areas (e.g., Louisiana in the US) (e.g. Terrell *et al.*, 2020), and assess $NO_2$ concentration differentials at the census tract level using instruments like TROPOMI (e.g. Hrycyna et al., 2022, Kerr et al., 2023).

Whilst the use of satellite data is relatively recent, improvements in spatial resolution, the range of considered pollutants, and accessibility to processed remote sensing data products are expected to drive increased usage (Gardner-Frolick et al., 2022, Sayyed et al., 2024). Satellite data, with consistent coverage of large spatial areas beyond regulatory monitoring networks, are valuable to explore associations with health outcomes and identify small communities that may be overburdened by air pollution (e.g., Demetillo et al*.,* 2020).

Although satellites have now demonstrated acceptable accuracy in estimating $PM_{2.5}$ and $NO_2$ concentrations on a small spatial scale (1 km x1 km), challenges persist, including limitations in measuring certain pollutants of concern (e.g., HAPs) due to optical measurement constraints. Satellites' data collection of total atmospheric column has challenges in then converting measurements to surface pollutant estimations, particularly for longer-lived air pollutants. Additionally, temporal coverage gaps are inherent as many satellites only measure once a day at specific locations and output may also be impacted by cloud cover (Gardner-Frolick et al., 2022).

**STATISTICAL METHODS**

Statistical approaches use monitor-based observations and show relationships providing estimates of spatial and temporal air pollutant concentrations and the uncertainties embedded in air quality data.

- *Geospatial interpolation methods (such as Kriging)*: This stands out for its utilization of advanced geostatistical techniques, assuming a Gaussian process. It is widely applied in air pollution and environmental justice studies (e.g., Nguyen et al., 2018; Xu et al., 2019) and health impacts (e.g. Jerrett et al., 2005) as it enhances accuracy in concentration estimates by considering spatial autocorrelation. It is relatively simple to implement with geographic information system programmes, interpretation is complex for pollutants with high variability and accuracy depends on a representative monitoring network. However, achieving a sufficiently dense network for correlated measurements can pose challenges in certain areas.

- *Land Use Regression (LUR)*: Land Use Regression estimates air pollution concentrations by relating land use variables with monitored data and is particularly effective for pollutants linked to land use type, for example traffic-related air pollutants (e.g. $NO_2$, $PM_{2.5}$ and BC). In addition, it is commonly applied in urban or regional studies (e.g., Clark et al., 2017, Knibbs et al., 2015) as LUR improves accuracy at finer spatial scales and adapts well to various monitoring schemes. Nevertheless, it requires access to geospatial variables which limits its applicability in areas with sparse data and is specific to pollutants associated with land use (Gardner-Frolick et al., 2022). LUR can however be employed over large areas and potentially back in time since detailed records of land-use change exist.

- *Hybrid models with machine learning*: These methods identify generalizable patterns offering potential applications in identifying links between air quality and other variables. While historically used for criteria pollutants and air quality indices, machine learning application in evaluating environmental differentials remains limited (Bellinger *et al*., 2017). Several studies have integrated it with non-regulatory monitors and satellite measurements to estimate neighbourhood-level concentrations (e.g. Zhang et al., 2020). Challenges include the complexity, opacity, and interpretability of results. As this method evolves its capacity to handle diverse data sets and provide increasingly accurate estimations may drive its broader adoption in air pollution research (Gardner-Frolick et al., 2022).

**PROCESS-BASED METHODS**

Numerous techniques for estimating air pollution concentrations involve modelling atmospheric processes such as advection, diffusion, and chemical reactions. These methods provide an approximation of how air pollution emissions are transported and disperse in the atmosphere, offering valuable insights into the fate and transport of these pollutants.

- *Dispersion models:* These models are based on atmospheric advection-diffusion equations with limited atmospheric chemistry and are valuable tools in estimating air pollution concentrations, if the pollutants are not highly chemically reactive. They are frequently employed in environmental studies and have been instrumental in assessing a wide range of impacts, such as the influence of vehicle emissions on schoolchildren (e.g., Batterman et al., 2014).

- *Chemical Transport Models (CTMs).* CTMs simulate atmospheric conditions, and chemical species using emissions and meteorological data (Gardner-Frolick et al., 2022). A wide range of pollutants can be represented with CTMs as they have the capability to simulate many processes, including photochemistry and chemical reactions. CTMs (e.g. GEOS-Chem, CMAQ, EMEP) are commonly used in environmental studies, especially for $PM_{2.5}$, $NO_x$ and $O_3$ (e.g. Bravo et al., 2016). These models enable large-scale analyses but they require significant computational resources.

**ECOLOGICAL AND AREA-BASED METHODS**

Many studies in the UK and EU have used ecological (or area-based) analysis of air pollutant concentrations against population socioeconomic or demographic variables to demonstrate differentials in exposure (e.g. Barnes et al., 2019; Brunt et al., 2017; Horton et al., 2023; Fecht et al., 2015; Mitchell and Dorling, 2003). These studies often use area- or population-weighted averages derived from gridded modelled concentrations. These are then applied to administrative or census geographic units, e.g. NUTS2 (Nomenclature of Terratorial Units for Statistics, county level for England) or LSOAs (Lower layer Super Output Area), based on the available spatial resolution of the socioeconomic or demographic variables selected. These approachese use place of residence as the location for exposure. Area-based analyses are limited as they do not represent the variability of concentrations, socioeconomic and demographic variables within each area and their accuracy depends on the spatial resolution of the study geography. Furthermore, utilising gridded modelled background concentrations does not present an accurate assessment of local exposure. Care must be taken in interpreting these studies to avoid the ecological fallacy, i.e. applying the findings at an area level to the individuals or households within. They may also be subject to the modifiable areal unit problem (MAUP) depending on the zonal design and spatial resolution of the spatial unit. (MAUP is a statistical bias that can occur during spatial analysis that causes differing results although the same analysis is applied to the same data.)

## ALTERNATIVE HYBRID METHODS

Hybrid approaches are frequently used to address the inherent trade-offs between different air pollution estimation techniques, reconciling the balance between computational intensity and concentration estimate accuracy (Gardner-Frolick et al*.,* 2022). The combination of methods provides a robust alternative, which requires careful consideration to ensure their relevance for the specific analysis in question. Many studies have effectively employed hybrid methods in air pollution and inequalities research, e.g., integration of interpolation and CTMs (e.g., Nguyen et al., 2018), interpolation coupled with dispersion modelling (e.g., Tayarani et al., 2016), and the combination of CTM with satellite observations (Kerr et al., 2023).

While individual methods have their inherent limitations, combining them strategically can yield more robust outcomes. For instance, studies on $NO_2$ trends and pollution-attributable health effects in the United States have historically relied on *in situ* monitors, leaving about 70% of urban areas unmonitored (Kerr et al., 2023). In the UK, the ground-level $PM_{2.5}$ monitoring network, mainly managed by the national AURN and local authorities, is heavily concentrated in urban regions, with over 95% of monitoring stations located in these areas (AQEG, 2024a). Novel tools, including satellite $NO_2$ and $PM_{2.5}$ observations and estimates from land-use regression and CTM, offer complete spatial coverage, providing a more comprehensive understanding of exposure gradients. In Kerr et al*.* (2023), the integration of *in situ* monitoring, satellite data, LUR, and CTM revealed ethno-racial disparities in $NO_2$ exposure in 10 main US cities (Figure 2.1). Black, Hispanic, Asian, and multiracial populations were found to experience $NO_2$ concentrations 15–50% higher than the national average, contrasting with the consistently lower exposure (5–15% below the national average) for the non-Hispanic White population. This study highlighted the importance of using diverse methods to capture a fuller picture of air pollution inequalities and the need for validation when using several methods.

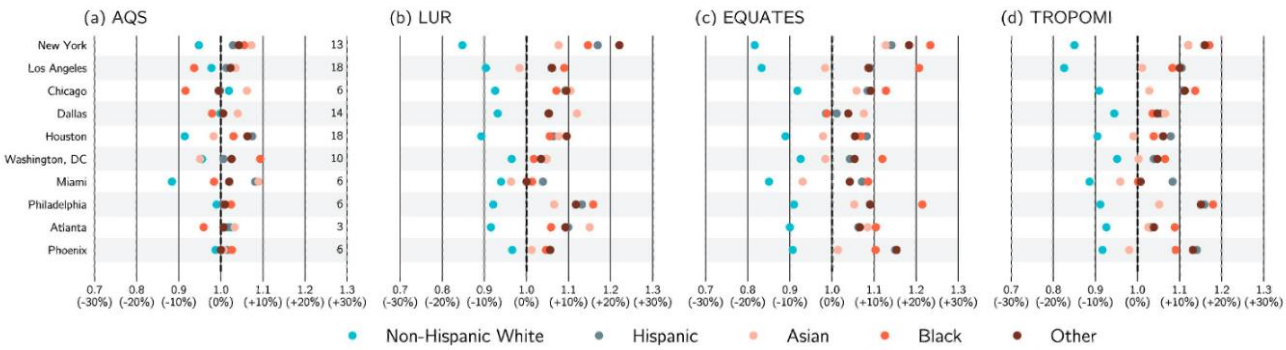


*Figure 2.1 Relative $NO_2$ disparities in main US cities calculated using (a) the nearest ground monitor to each census tract and tract-averaged $NO_2$ (AQS) (b) land use regression (LUR), (c) photochemical model (EQUATES), and (d) satellite observations (TROPOMI). The dashed black vertical line highlights a value of 1, which indicates that a particular subgroup has the same $NO_2$ concentrations as the overall population-weighted average for a given city. Numbers in (a) represent the number of AQS monitors in each city, adapted from Kerr et al., (2023).*

### 2.2.2 Qualitative Approaches

Qualitative methods play an important role in comprehending the impact of air pollution on individuals, relying on qualitative data derived from interviews, focus groups, and surveys (Abed Al Ahad et al., 2023; Scammell, 2010). Among these methods, community-based participatory research stands out, emphasizing collaboration with community members to identify research questions, collect data, and interpret findings. This inclusive approach fosters a deeper understanding of local experiences and perspectives (Larsson et al., 2006, Moody et al., 2021; Fogg-Rogers et al., 2024). Another significant qualitative method is environmental justice storytelling, which employs stories and narratives to underscore the disproportionate impacts of environmental hazards on marginalized communities (e.g., Peres et al., 2006, Wenham, 2007). As well as determining exposure effects, qualitative methods can be used to evaluate differences between social and economic groups' attitudes to air quality policies (Mebrahtu et al., 2023). There is evidence to suggest that inequitable access to air quality data may also result in differential exposures as some sectors of society may be more well-informed and hence take evasive action (Schulte, 2022; Schulte and Hudson, 2023).

## 2.3. Methods and data resources in the UK

Much of the research literature identified in Section 2.2 was in a US context, in part a reflection of origins of environmental justice as a concept, however there are also a number of UK studies. Whilst US studies have historically focused on ethnicity, the US Government definition of environmental justice now extends to everyone, regardless of income, race, national origin, Tribal affiliation, or disability. It also covers cumulative impacts arising from environmental and other burdens, including systemic barriers, to ensure equitable access to healthy, sustainable and resilient environments (USEPA, Executive Order 14096). A series of Federal Executive Orders in 2021 and 2023 built on President Clinton's 1994 EO 12898, to raise the status of environmental justice to a "whole of Government" approach". Tools such as EJScreen bring together a wide range of environmental indicators (including criteria air pollutants and toxics), socioeconomic indicators (including income, unemployment, education and age), with health disparities (e.g. asthma and heart disease), climate risks (e.g. flooding and wildfires) and critical service gaps (e.g. housing burden and transportation access) to more holistically address environmental inequalities.

In the UK, although there have been a small number of research studies covering ethnicity associations and air pollution, research has largely focused on the relationships between specific environmental and socioeconomic factors. There has historically been less emphasis on holistic screening and assessment tools; however, there are some recent developments that attempt to address this:

- SHAPE Place is an interactive data mapping, analysis and insight tool covering England, supported by the Department of Health and Social Care. It is free to use for the public sector and intended to support service and estate planning, although has the potential to be a valuable research tool as well. UKHSA has developed a PILOT indicator to represent population level vulnerability to air pollution at LSOA level. This is a ranking of the level of vulnerability from low (1-2) to high (9-10) decile scores. It is based on the population characteristics (% of young people (<16 years) and older adults (65+ years)), Levels of Deprivation (Index of multiple deprivation score), location of vulnerable populations (hospitals, schools, care homes and childcare facilities) and the concentration of air pollution (NO2 and $PM_{2.5}$) modelled for 2018. The tool also includes several other environmental, demographic and access/usage layers, for various administrative areas, giving the potential for a more systematic and integrated appraisal of risk.

- Ambient air quality ($NO_2$, $PM_{10}$, $SO_2$) forms part of the Access to Healthy Assets & Hazards (AHAH) Index (v.3), alongside Retail environment (access to fast food outlets, pubs, tobacconists, gambling outlets), Health services (access to GPs, hospitals, pharmacies, dentists, leisure services), and Physical environment (such as Blue Space, Passive Green Space definitions). AHAH is a multi-dimensional index developed by the Consumer Data Research Centre (CDRC) for Great Britain measuring how 'healthy' neighbourhoods are. Sheffield City Council's Local Insight (communityinsight.org) provides a platform for local interpretation of the data. The future of the AHAH may be limited however, subject to funding.

The UK has many comprehensive open-access government datasets including detailed emissions inventories, extensive monitoring networks and detailed demographic census information, which provides opportunities for the identification of associations and the disaggregation of possible confounding effects.

# Chapter 3 – Spatial variability of emissions and concentrations

Distributional differentials associated with air quality are driven by the spatial variability of air pollutants emissions detrimental to human health and where people and different demographic groups spend their time in relation to this varying concentration field. For this assessment, it is useful to examine the spatial pattern of emissions and the behaviour of pollutants from the point of emission to their final environmental fate. Such considerations can help to understand the important underlying factors that affect the distribution of pollutants in the atmosphere and how that might be related to differentials in individual air pollutant impacts.

The location and nature of emission sources is a major determinant of exposure to air pollution and is often used as a proxy. However, using the distribution of emission sources alone as an indicator of exposure to pollutant concentrations will not capture potentially important influences related to pollutant dispersion and the regional background concentration. The ground-level impact of emissions from tall chimney stacks will be very different to emissions released close to the surface for example and precursor emissions released at one location will generate secondary pollutants, such as $PM_{2.5}$ organic aerosols, which expose populations over large areas. Air quality modelling is therefore required to understand impacts and distributions of concentrations. Related to this point is the environment into which emissions are released. While proximity to a source is of primary importance, so too is the influence of the environment itself. For instance, the influence of street canyons compared with more open locations, where the former would tend to result in accumulation and higher concentrations than the latter for the same amount of emission.

The geographic scale over which inequalities are considered is an important factor which is influenced by the behaviour of individual pollutants in the atmosphere. For primary pollutants, the influence of dispersion and proximity to a source are of central importance. For pollutants such as $NO_x$ and $NO_2$ there can be steep gradients in concentrations with short distances down to a few metres. However, for many secondary pollutants (pollutants that are formed in the atmosphere through chemical reactions), or pollutants that have a large natural or secondary component, such as $PM_{2.5}$, the gradients in concentration at a country scale will be important.

## 3.1 Distribution of ambient $NO_2$ concentrations

The distribution in annual mean ambient concentrations of $NO_2$ is shown in Figure 3.1 (left panel, for year 2019), illustrating that there is a range of concentrations experienced across the UK from < 5 $\mu g\ m^{-3}$ in northern Scotland and rural parts of Northern England and Wales, to in excess of 40 $\mu g\ m^{-3}$ close to roads in London, and with elevated concentrations in other towns and cities and the trunk roads between them. An example centred around the Manchester area (Figure 3.1 right panel), shows a highly structured distribution of $NO_2$ concentrations typified by complex array of road traffic sources, superimposed on other significant $NO_x$ sources (for example Manchester airport towards the lower part of the image).

Source apportionment of $NO_2$ for the four UK regions is shown in Figure 3.2; road transport is currently a major source, as well as non-road (e.g. airports), shipping and large industry and domestic combustion. The short atmospheric lifetime of $NO_2$ means that longer range transport has a more limited role to play in exposure, making it a largely locally controllable pollutant.

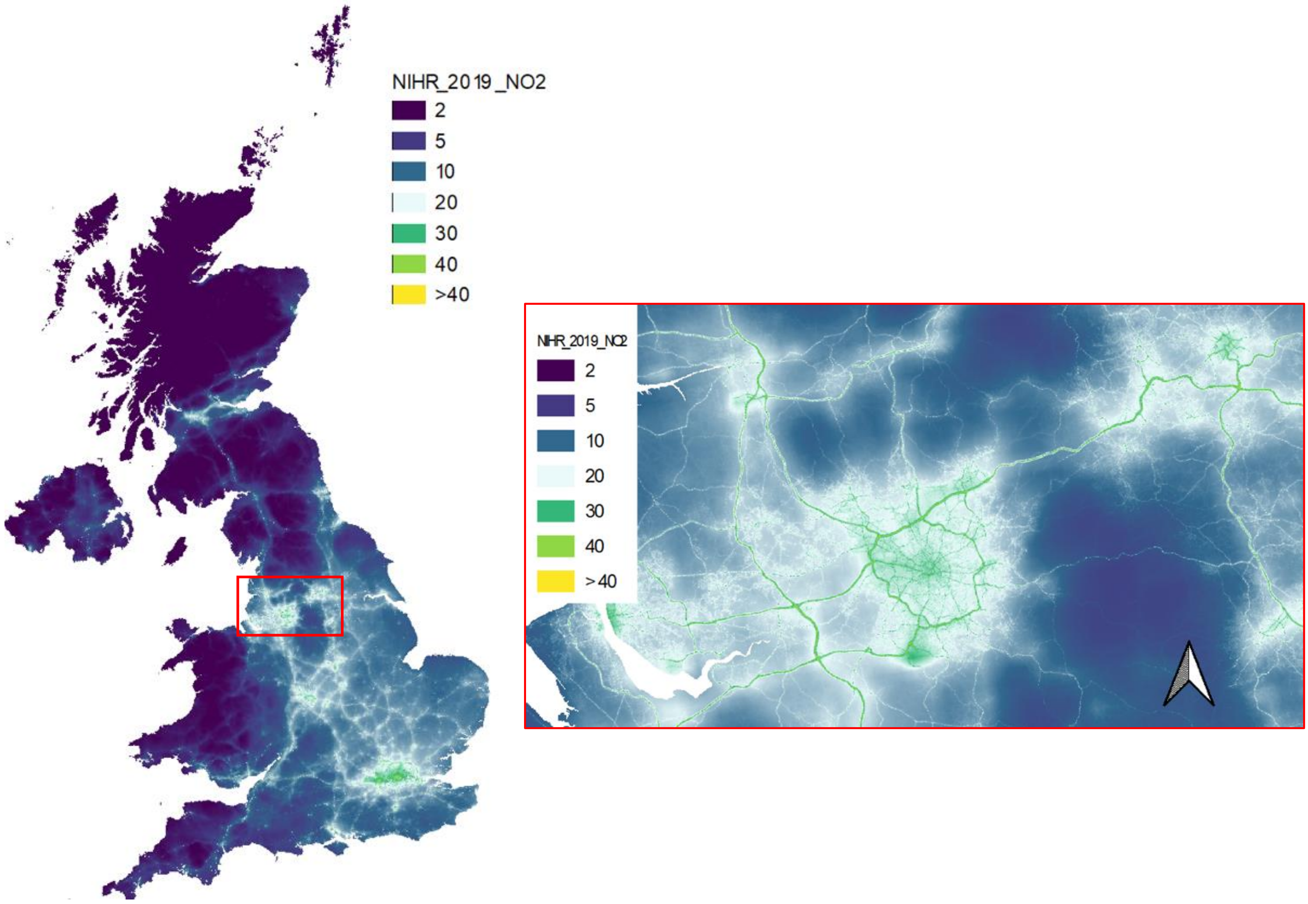


*Figure 3.1. Model estimated annual average $NO_2$ concentrations ($\mu g\ m^{-3}$) for 2019, taken from a coupled CMAQ/ADMS model at 20 m resolution close to roads. Top- UK-wide distribution. Right – inset red box section for Manchester and broader northern England region.*

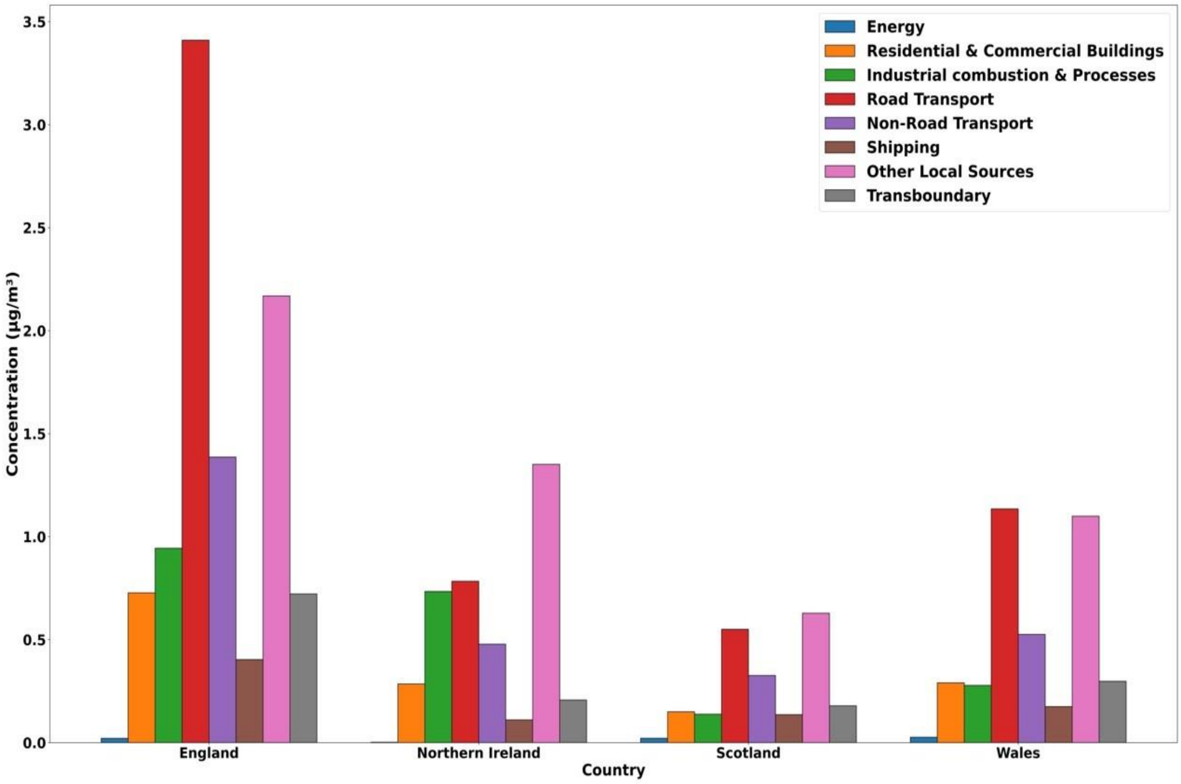


*Figure 3.2. Contributory sources by emission sector to the England, Northern Ireland, Scotland and Wales annual average $NO_2$ $\mu g\ m^{-3}$ for 2019. Data from the Integrated Source Apportionment Method (ISAM) version of CMAQ. 'Other local sources': includes SNAP6 (Solvent use), SNAP9 (Waste treatment), SNAP10 (Agriculture), SNAP11(Natural/Biogenic) and large point sources. Transboundary includes the contribution from sources outside UK.*

Average roadside $NO_2$ concentrations in England have fallen considerably from 59.7 µg $m^{-3}$ in 1997 to 24.7 $\mu g\ m^{-3}$ in 2021, a decrease of ~59% over the latest 24 years for which data are available (Figure 3.3). The general trend in measured ambient $NO_2$ concentrations is a decreasing one and the average ambient concentrations has fallen below the current $NO_2$ limit value of 40 $\mu g\ m^{-3}$ in recent years. However, there has been a slight increase in average $NO_2$ concentrations following the lifting of COVID-19 travel restrictions as the time series reached its lowest point in 2020 because of an unusually low level of road traffic.

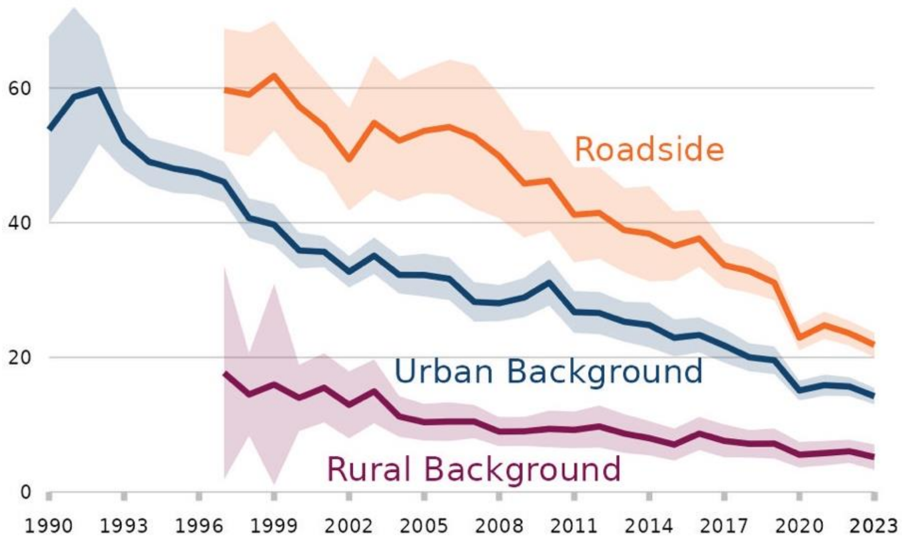


*Figure 3.3. Trends in average roadside, urban background and rural background $NO_2$ concentrations in England (1990-2023). Department for Environment, Food & Rural Affairs, ENV02 - Air quality statistics https://www.gov.uk/government/statistics/air-quality-statistics/ntrogen-dioxide  Unit of Measurement Annual mean ($\mu g\ m^{-3}$). The shaded area in the graph represents the 95% confidence interval (measure of uncertainty) for the annual mean concentration measured at roadside sites. The interval narrows over time because of an increase in the number of monitoring sites and a reduction in the variation between annual means for $NO_2$.*

## 3.2 Distribution of ambient $PM_{2.5}$ concentrations

$PM_{2.5}$ has a relatively long atmospheric lifetime (several days), and both primary and secondary sources, so the differentials between areas are less distinct than for $NO_2$. $PM_{2.5}$ accumulates as air passes over the UK, so the geographic location of individual cities becomes significant since the amount of $PM_{2.5}$ in each city is impacted by upwind sources. For example, large urban areas such as Glasgow and Edinburgh have lower ambient $PM_{2.5}$ than comparably sized urban areas of Manchester and Birmingham further south (Figure 3.4). This is because the local city emissions of $PM_{2.5}$ are added to a very low rural background concentration of $PM_{2.5}$ across most of Scotland.

This underlying large-scale gradient in background concentrations is strongly influenced by the distance from other sources on mainland Europe, shipping in the English Channel, and larger population densities in the southeast causing enhanced regional emissions. There are further meteorological effects with higher rainfall and windspeeds in the north of the UK, which leads to greater $PM_{2.5}$ washout and dilution. Within an individual urban area, the gradients in long-term average concentrations will tend to be determined by local primary sources that are superimposed on top of background secondary pollutants. However, the UK north-south difference in concentrations of $PM_{2.5}$ may have a more significant effect on differentials in air pollution than intra-urban differences. According to Woodward *et al*. (2024), the bias towards more deprived areas in England for total $PM_{2.5}$ is greater when considering $PM_{2.5}$ from UK anthropogenic sources only, which suggests that “evaluating the bias in exposure across deprivation deciles using total $PM_{2.5}$, without considering source-apportionment, does not provide an accurate assessment of the impact of UK emissions on the bias in exposure".

Figure 3.4 (left) shows that there are a wide range of annual average $PM_{2.5}$ concentrations experienced across the UK ranging from approximately 3 $\mu g\ m^{-3}$ in Northern Scotland and rural parts of Northern England and Wales, to in excess of 11 $\mu g\ m^{-3}$ close to roads in London and in other towns and cities. For the example of the area around Leeds and Manchester (Figure 3.4 (right)) the picture is similar to $NO_2$, but with a relatively smaller road traffic influence. Another important factor from source apportionment of $PM_{2.5}$ (Figure 3.5) is that transboundary sources outside the UK (hemispheric, European and shipping) alongside sea salt, and other local UK industry, agriculture, biogenic and mineral sources as well as secondary organic aerosols are significant contributors and vary more smoothly from high concentrations in the southeast reducing towards the northwest.

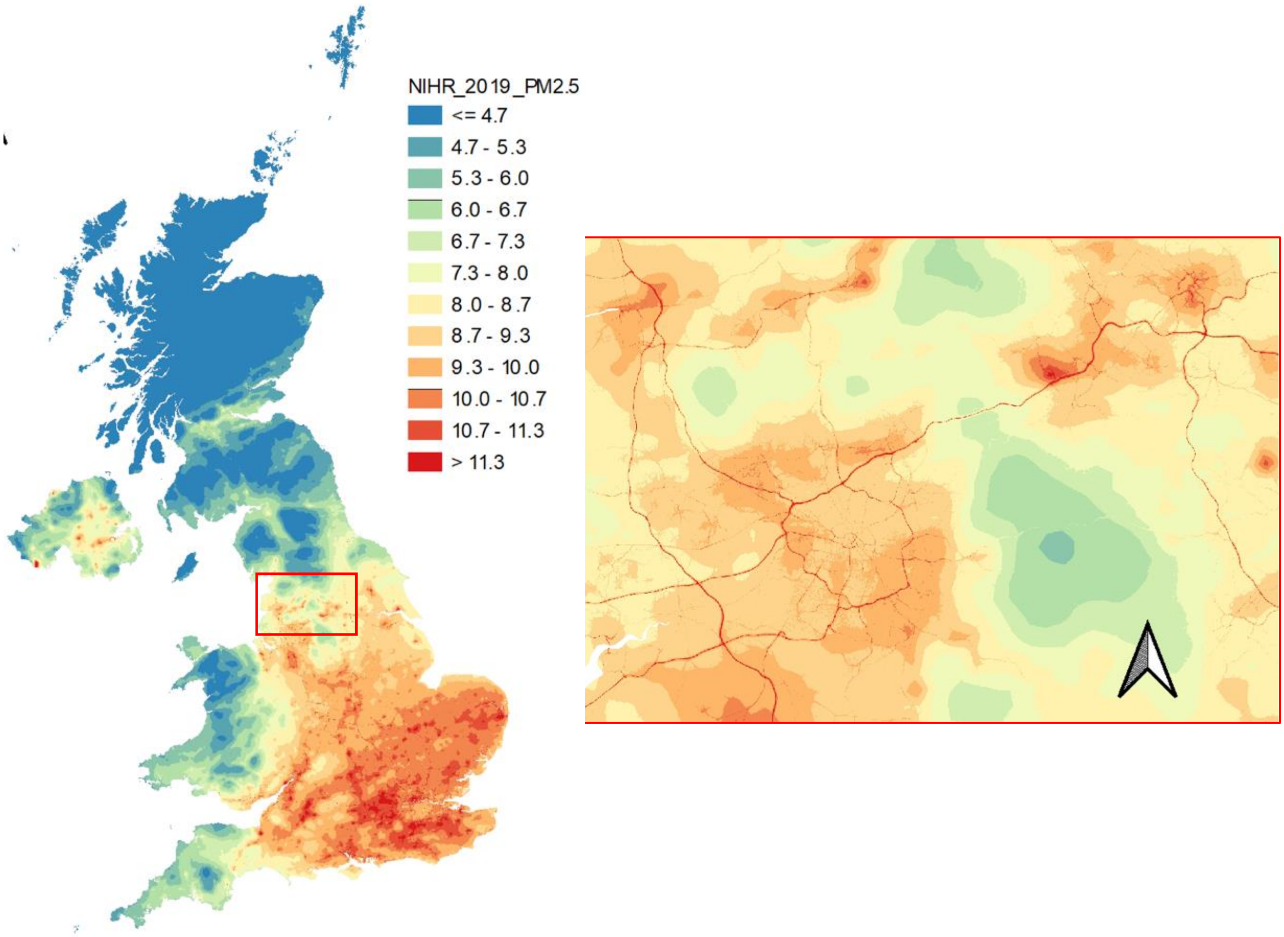


*Figure 3.4. Left: Annual average distribution of $PM_{2.5}$ µg $m^{-3}$ across the UK in 2019 taken from a coupled CMAQ/ADMS model at 20m resolution close to roads. Right: Annual average distribution of $PM_{2.5}$ µg $m^{-3}$ across the Liverpool – Manchester – Leeds northern England region taken from a combined CMAQ/ADMS model at 20m resolution close to roads.*

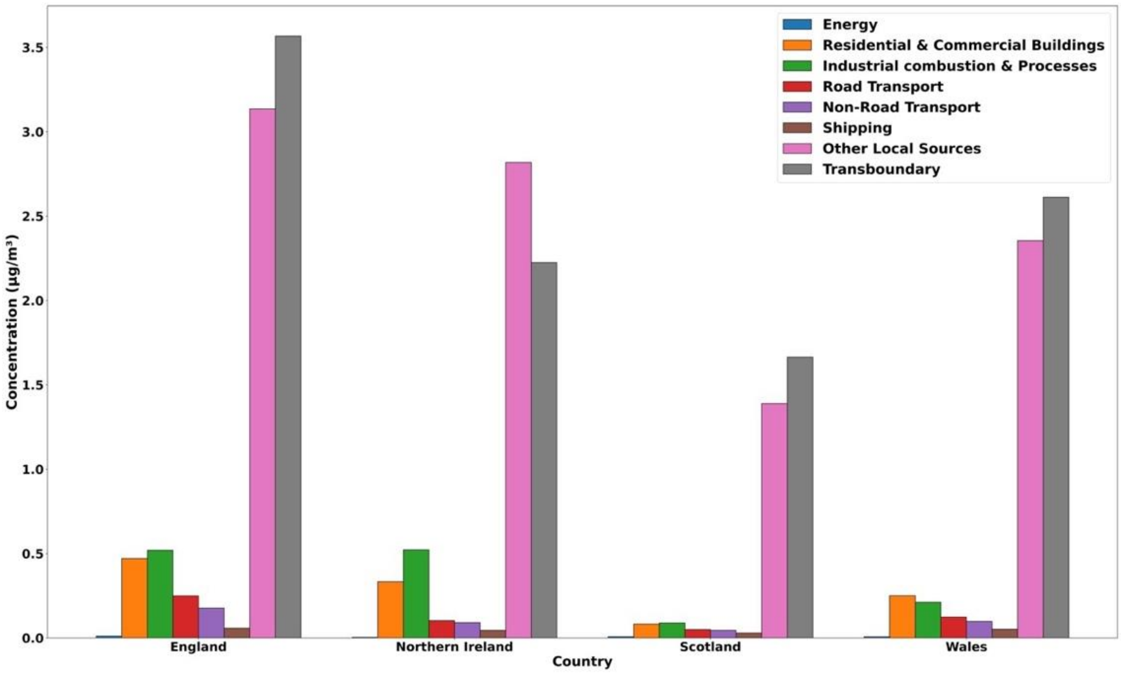


*Figure 3.5. Contributory sources by emission sector to the England, Northern Ireland, Scotland and Wales annual average $PM_{2.5}$ µg $m^{-3}$ for 2019. Data from the Integrated Source Apportionment Method (ISAM) version of CMAQ. 'Other local sources': includes UK minerals (Si, Fe, Ni, Mg, Mn, and Al), SNAP6 (Solvent use), SNAP9 (Waste treatment), SNAP10 (Agriculture), SNAP11(Natural/Biogenic), large point sources and secondary organic aerosols. Transboundary includes the contribution from sources outside the UK.*

Defra-reported statistics are for a population-weighted annual mean concentrations of $PM_{2.5}$ in England that has fallen from 12.1 µg $m^{-3}$ in 2011 to 7.4 µg $m^{-3}$ in 2021, a decrease of 39% over the latest 10 years for which data are available. Figure 3.6 shows this dataset extended to 2022, of note are the reductions in $PM_{2.5}$ since in 2020 and 2021 associated with COVID-19 activity restrictions.

$PM_{2.5}$ annual mean concentrations are estimated annually for every square km of the UK using the Pollution Climate Mapping (PCM) model. The geographical distribution of the UK population is then joined to the estimated concentrations to estimate the annual mean concentration of $PM_{2.5}$, weighted by where the population lives. This accounts for most of the population living in densely populated urban areas, where concentrations are likely to be greatest.

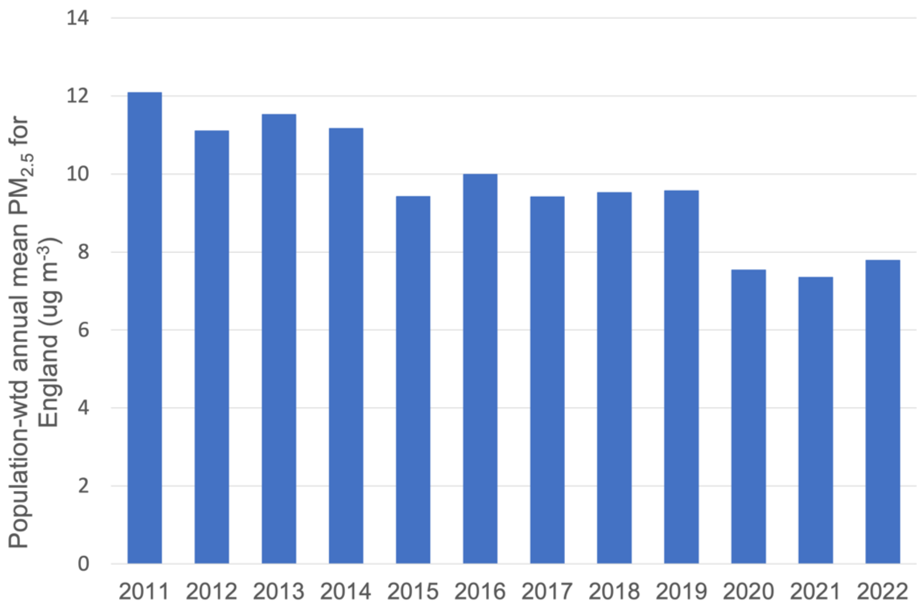


*Figure 3.6. Population weighted annual mean $PM_{2.5}$ concentrations* (μg $m^{-3}$) *for England calculated using the Pollution Climate Model (PCM). Note: Concentrations of $PM_{2.5}$ vary from year-to-year due to the weather. This variation due to weather is generally greater than the year-to-year variation from changes in emissions*

## 3.3 Distribution of ambient $O_3$ concentrations

The distribution of concentrations of $O_3$ across the UK is shown in Figure 3.7 (left panel) for 2019. This illustrates an inverse distribution in concentrations across the UK compared with $NO_2$ and $PM_{2.5}$, from greater than 50 μg $m^{-3}$ in Northern Scotland and rural parts of Northern England and Wales, to less than 26 μg $m^{-3}$ close to major roads, around Heathrow in London and in other towns, notably around Southampton, the Midlands, Manchester and Liverpool. Emissions of NO react rapidly with $O_3$ to form $NO_2$ and as a result, $O_3$ also shows a very detailed structure in cities, similar to $NO_2$ but with opposite spatial trends, with roadsides having a minimum in exposure, down to ~10 - 15 μg $m^{-3}$, urban background concentrations being higher and rural concentrations being higher still. This is demonstrated by the map of $O_3$ in London in Figure 3.7 (right panel) and the associated transect of $O_3$ and $NO_2$ concentrations (Figure 3.8), which show the $O_3$ minimum concentrations at roadsides in central London, and the opposite highest concentrations for $NO_2$.

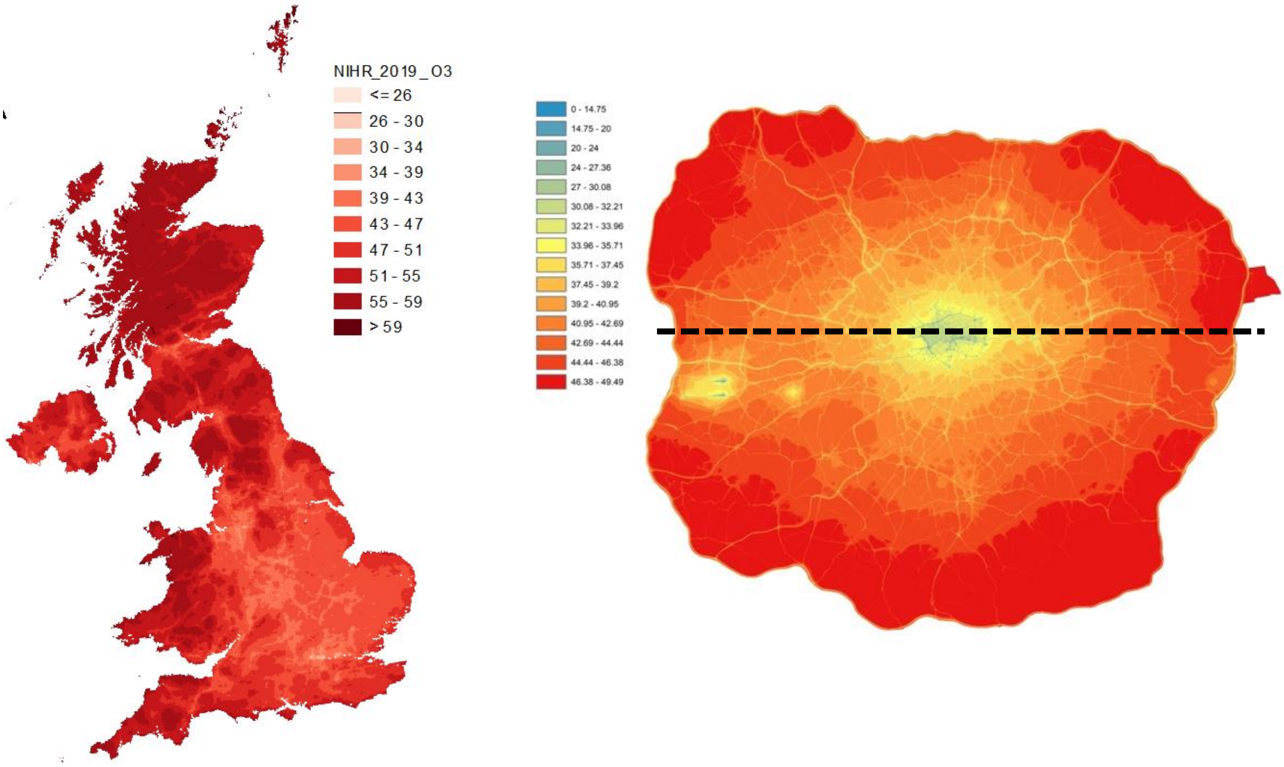


*Figure 3.7. UK annual average $O_3$ μg $m^{-3}$ for 2019 (left) taken from the CMAQ model (Byun et al. 2006) at 2 km resolution, and in London – defined by M25 boundary (right) at 20 m resolution, based on the London Toolkit model. (Dajnak et al. 2023).*

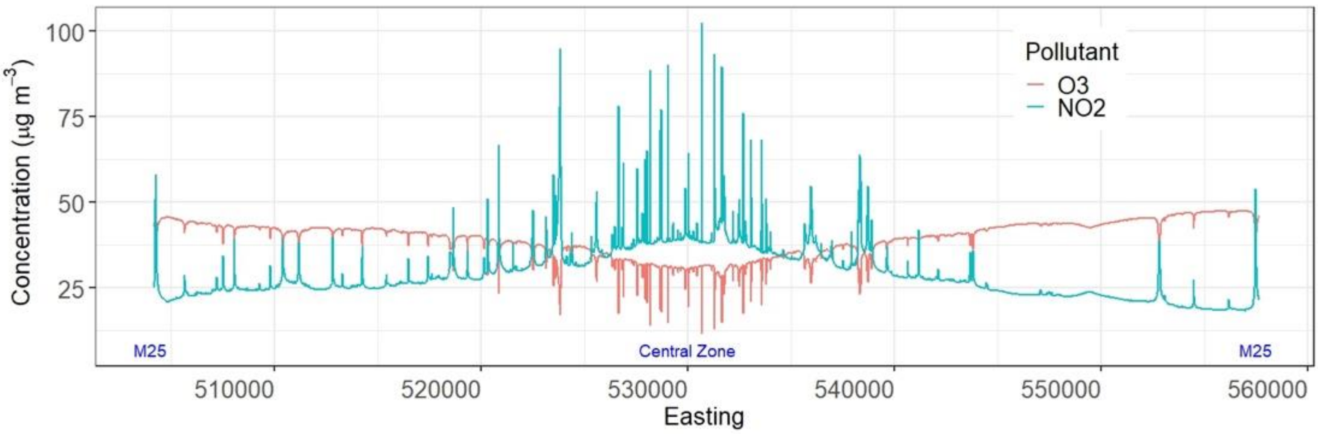


*Figure 3.8 The concentration of $O_3$ and $NO_2$ along the black dashed transect line across London (shown in Figure 3.7).*

To assess the degree to which differentials in emission distributions are similar to, or differ from, concentration patterns, the general relationship between emissions and concentrations can be explored by comparing the NAEI 1 km x 1 km emissions total with the 1 km x 1 km modelled concentration for the same pollutant. The correspondence between $NO_x$ emissions and $NO_2$ concentrations is shown in Figure 3.9 where 1 km x 1 km means are compared across the UK for 2021. Figure 3.9 shows that while there is a general tendency for $NO_2$ concentrations to increase with $NO_x$ emissions, there is a very large variation in grid square $NO_2$ concentrations for a given specific value of $NO_x$ emission in the same grid box. Much of the reason for such a large spread is caused by the way in which emissions are released e.g. ground level sources such as road transport and elevated stack emissions related to power generation, as well as the effects of atmospheric chemistry and meteorology on pollutant formation and dispersal.

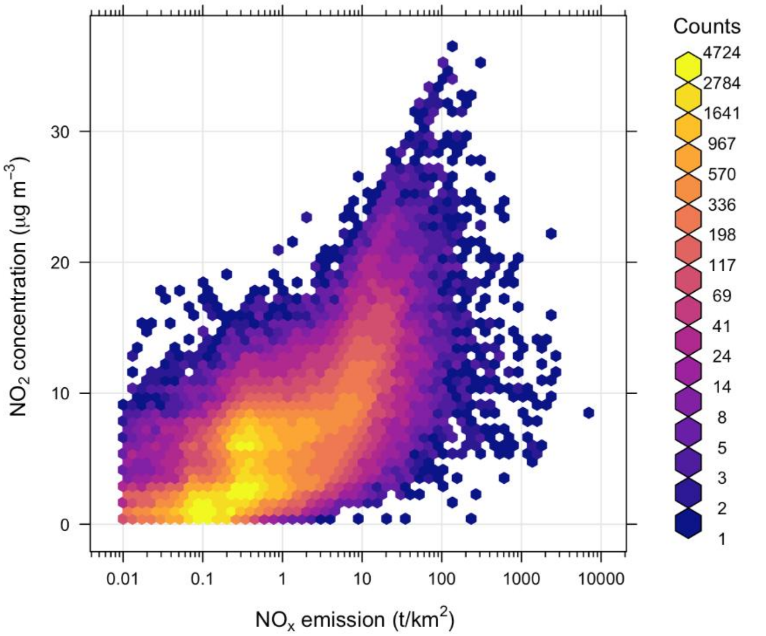


*Figure 3.9. Relationship between NAEI 1 km x 1 km $NO_x$ emissions and corresponding 1 km x 1 km modelled annual mean concentrations of $NO_2$ for 2021. Note the log scale for emissions.*

Similarly, the relationship between primary $PM_{2.5}$ emissions and concentrations (Figure 3.10) shows rather little correspondence between primary emissions and modelled ambient concentrations, with any specific level of direct emission being associated with a very wide range of possible ambient concentrations experienced. This is not an unexpected outcome since so much of the $PM_{2.5}$ concentration is derived remotely from point of observation.

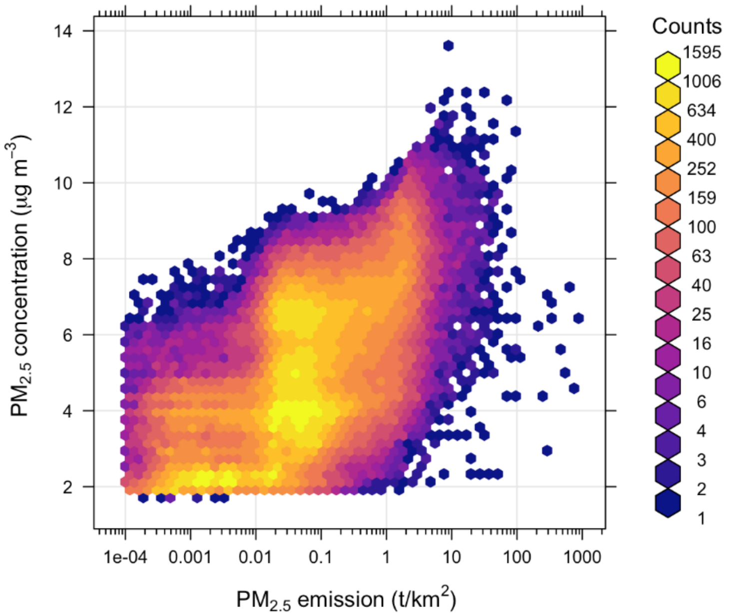

*Figure 3.10. Relationship between NAEI 1 km x 1 km $PM_{2.5}$ emissions and corresponding 1 km x 1 km modelled annual mean concentrations of $PM_{2.5}$ for 2021. Note the log scale for emissions.*

Another way to consider differences in the spatial variation in emissions and concentrations of air pollution is to consider a transect (slice) through the UK. Figure 3.11 shows a west-east transect for $NO_x$ emissions and $NO_2$ concentrations that passes through central London. This figure uses the PCM model to estimate both emissions and concentrations of $NO_x$ and $PM_{2.5}$. Figure 3.11 reveals (as Figure 3.9) that there is a reasonable correspondence between emissions and concentrations but also locations of very high emissions where there is no obvious effect on localised $NO_2$ concentration. Indeed, the highest emission in a 1 km x 1 km grid box (off the scale) in Figure 3.11 is 833 tonnes $yr^{-1}$.

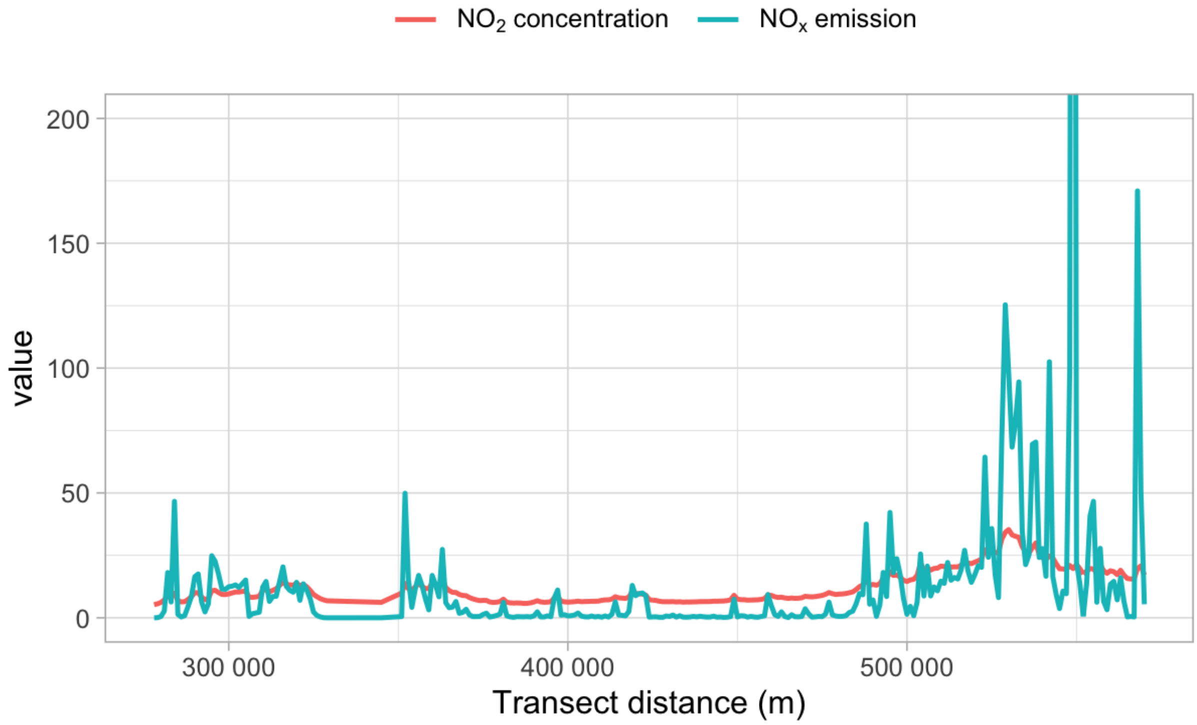


*Figure 3.11. West to East transect of NAEI 1 km x 1 km $NO_x$ emissions (y-axis Value is in units of tonnes $yr^{-1}$) and corresponding 1 km x 1 km modelled annual mean concentrations of (y axis value is in µg $m^{-3}$) for 2021. The transect is chosen to cross central London in a west-east direction.*

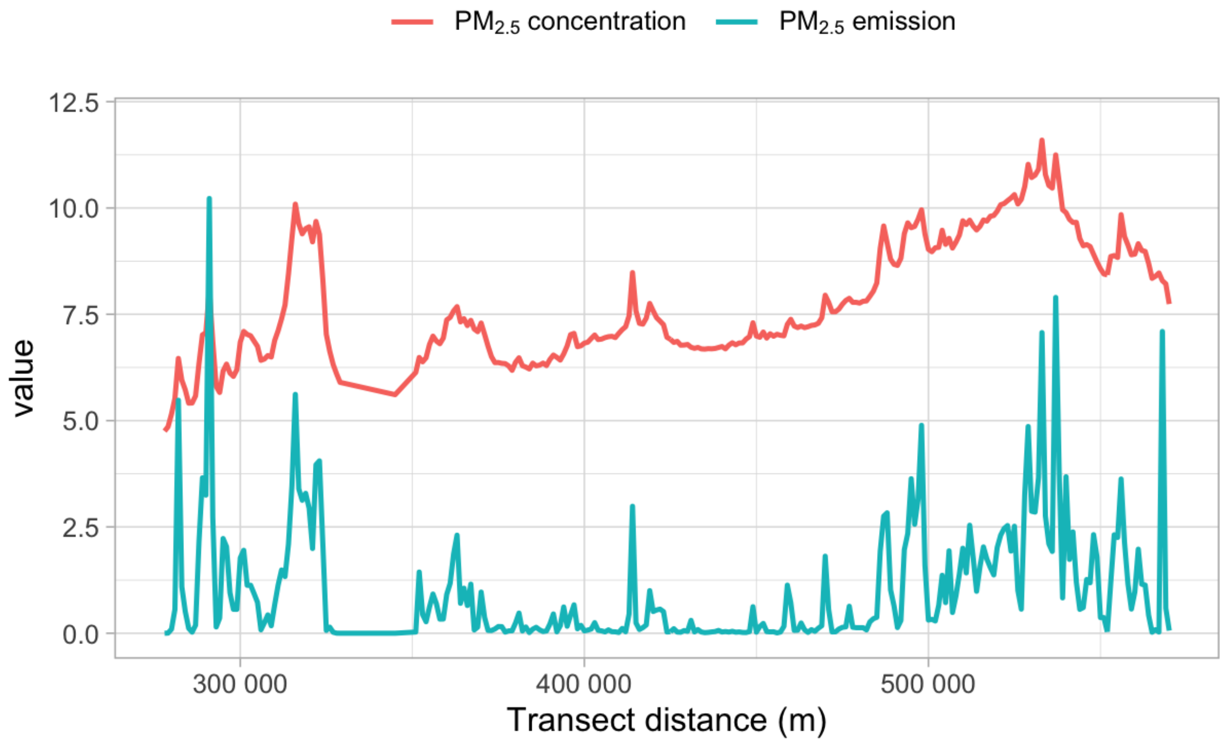


*Figure 3.12. West to East transect of NAEI 1 km x 1 km $PM_{2.5}$ emissions (y-axis value is in units of tonnes $yr^{-1}$) and corresponding 1 km x 1 km modelled annual mean concentrations of (y-axis value is in units of µg $m^{-3}$) for 2021 The transect is chosen to cross central London in a west-east direction.*

The transect for $PM_{2.5}$ (Figure 3.12) highlights the overall important effect of background concentrations. While the influence of London can be seen on the right-hand side of the plot for both emissions and concentrations, there is a strong influence of a high regional background component across the transect. The generally increasing $PM_{2.5}$ concentration in going from west to east is also clear in Figure 3.12, again due to the influence of a regional influence in $PM_{2.5}$ concentrations that increases further east. Figures 3.11 and 3.12 consider emissions and concentrations at a 1 km resolution which is too coarse to reveal the detail of many sources such as the fall-off in concentration away from specific roads. However, these plots do emphasise the general importance of geographic variability in relating either emissions or concentrations to other factors that may indicate differentials in concentrations in different parts of the UK.

Related to the previous point on spatial variability is the temporal variation in concentrations and especially short-term air pollutant episodes. Many studies of air pollution differentials tend to consider long-term (typically annual) average concentrations as this relates to chronic exposure. Exposure to shorter-term high concentrations of pollutants will also vary depending on location. In this case there will likely be important differences between local, primary sources affected by local meteorology (important for $NO_2$ concentrations) compared with the influence of episodes dominated by transboundary air pollution (important for $PM_{2.5}$ concentrations).

A specific issue relates to assessment of $NO_2$ and associated differentials between locations. $NO_2$ and $O_3$ are closely coupled in the atmosphere, and it would generally be expected that locations with the highest $NO_2$ concentrations will have the lowest $O_3$ concentrations (and vice-versa). In that sense, the disparities related to $O_3$ exposure would tend to be opposite to $NO_2$. Indeed, in the case of $NO_2$ and $O_3$, considering the sum (known as the total oxidant, or $O_X$) might be a better indicator of health impacts (Williams et al., 2014). In general, considering a single pollutant in isolation might result in a potentially misleading assessment of the impact. Central to this issue is an understanding of the differential impacts that pollutants have on human health and their relative exposure-effect timescales, which remains uncertain.

Finally, there is the issue of actual personal exposure to air pollution. It is well-established that people spend the majority of their time indoors; often 80-90% of the time (AQEG, 2022). This has two implications: 1) total exposure may not be well represented by comparing against a static spatial correlation e.g. based on residential location, since people move around each day spending time commuting and in other locations, and; 2) indoor air quality may have more influence on exposure and thus health than outdoor air quality. For this reason, the geographical analysis above may not accurately represent the extent of difference associated with exposure.

# Chapter 4 – Differentials in primary emissions of $PM_{2.5}$ and $NO_x$ in the UK

Air pollutant emissions provide a metric to evaluate differentials in air pollution distribution. While differentials in pollutant emissions do not directly relate to differentials in concentration or exposure, there are benefits to using emissions data from both a technical and policy perspective. Highly detailed emissions data is available at 1 km x 1 km resolution across the whole of the UK which allows relatively easy comparison to socioeconomic data such as the Index of Multiple Deprivation, and to census information on demographics and ethnicity. Emissions data can be separated out into emissions by individual sources. Many government policies focus on reducing emissions from individual sectors and so actions that seek to reduce disparities in pollution exposure are largely focused on this metric of pollution. Considering where activities that produce air pollution emissions occur, and who is producing those emissions, allows issues of environmental justice to be explored.

## 4.1 Differentials in proximity to air pollution sources

Gray et al. (2023) considered deprivation-based inequalities in $NO_x$ emissions in England. The Index of Multiple Deprivation (IMD) was used as the deprivation indicator. The IMD and emissions data were linked by Lower Super Output Areas (LSOAs), geographical areas shaped so that they represent approximately 1600 people, which are typically between 0.1 and 2 $km^2$. Data was divided into deciles with 1 being the most deprived and 10 being the least deprived.

In England, geographies representing the least deprived decile experienced on average 44% less median annual $NO_x$ emissions compared to the most deprived decile, 55% less when the difference is calculated using a linear fit through the median points, and 57% less when calculated using linear regression of the entire data set. Figure 4.1 shows how emissions change by deprivation decile for different source sectors. Each source sector with annual emissions above 0.1 tonnes per $km^2$ has a reduction of at least 25% in emissions when going from the most to least deprived decile demonstrating that although road transport is a source of inequalities, other sectors display a similar trend in emissions by deprivation decile.

An important point to note here is that this analysis is all based on average values, but the emissions data has a significant skew so neither measure of central tendency (mean or median) is fully representative of the spread of the data. This has implications for the scale of inequality in the geographical distribution of emissions reported, as the average values will be considerably lower than the highest emissions experienced, and more deprived areas experience more high emitting outliers.

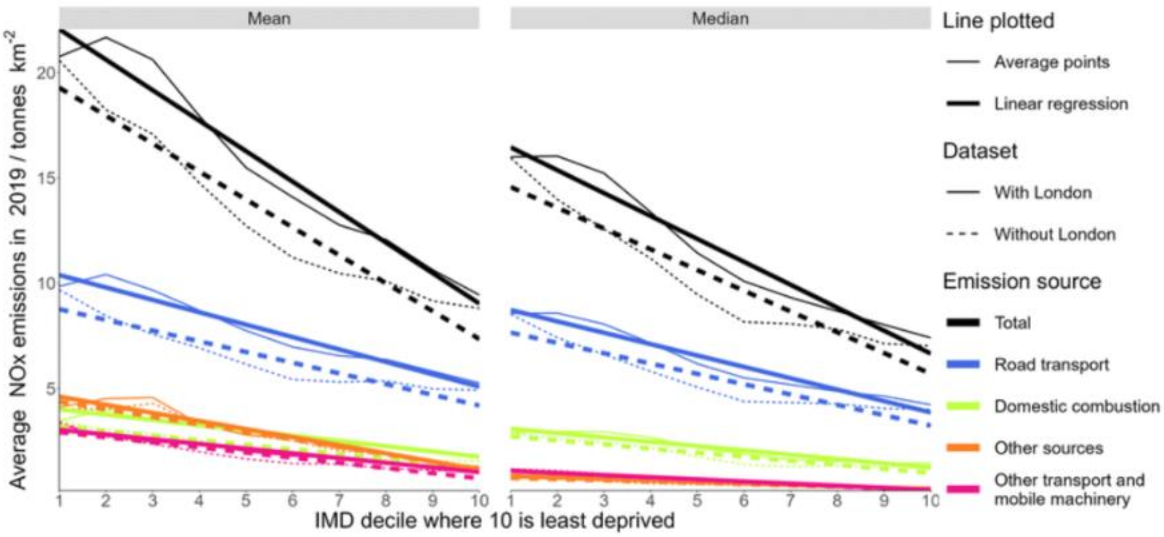


*Figure 4.1. The mean and median emissions of $NO_x$ by LSOA across England for major source sectors, the average without London is also shown. The category "Other sources" is the combination of the source sectors: agricultural; energy production; industrial combustion; solvents; natural; and point sources. From Gray et al. (2023).*

Calculating the gradient of emissions by deprivation decile produced by a linear model for each county or Unitary Authority (UA), it was estimated that at least 66% of counties or UAs in England have significant deprivation-based inequality in $NO_x$ emissions, and the true number is likely higher. It was also the case that when looking at average deprivation and emissions for each county and UA, the more deprived regions were more likely to experience higher emissions. The same was true for those cities where the variation in the trend in emissions by deprivation decile for cities was high, straight lines were less likely to be the best representation for the data. This suggests differences between cities exist in how emissions are distributed, so action to reduce disparities needs to consider the local situation.

Considering deprivation in different rural and urban classifications showed that there is a contrast between urban conurbations, which have high emissions and substantial areas of high deprivation, and rural towns and villages, which have low emissions and deprivation, which contributes to overall national inequality. However, it was also noted that those in more deprived areas are more likely to have relatively higher emissions of NOx regardless of whether they are in large urban cities or smaller, more rural towns. Perhaps surprisingly it was also seen that in larger LSOAs, i.e. those with lower population density, there was greater deprivation-based emission inequality than in smaller LSOAs (Figure 4.2). This appeared to be driven by emissions from point sources, whilst for smaller LSOAs there was a greater contribution from transport and other sectors.

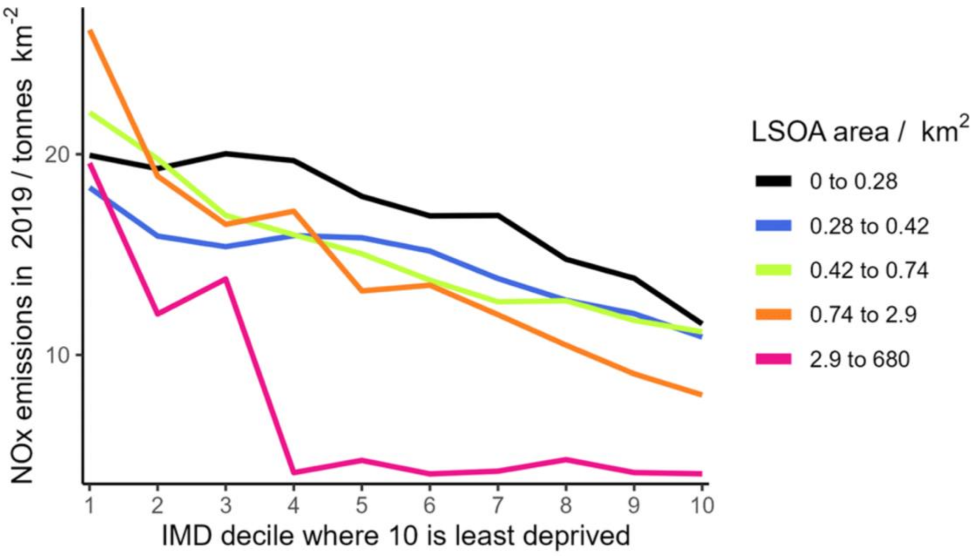


*Figure 4.2. Inequality in mean $NO_x$ emissions for different sizes of LSOA. The division was made purely on the expanse of each LSOA so each size bracket does not contain an even distribution of deprivation deciles. From Gray et al. (2023).*

The relationship between proximity to emission sources and ethnicity in England was explored by Gray et al., 2024. Similar trends were observed as found in concentration-based analyses that is, as the fraction of minoritised ethnicity population increased in a location, air pollution emissions of $NO_x$ and $PM_{2.5}$ also increased

This analysis took advantage of the high-resolution data on emissions and ethnicity available for England, and the level of detail around ethnicity collected which allowed sub-groups of typically used broad ethnic groupings (e.g. Asian) to be explored. All 24 minoritized ethnic groups studied experienced higher average local $NO_x$ and $PM_{2.5}$ emissions than socioeconomically matched populations in the majority “White: English, Welsh, Scottish, Northern Irish or British” ethnic group. The broad groupings hide the variation in both average level of deprivation and emissions close to place of residence as can be seen in Figures 4.3 and 4.4. The deprivation for the broad ‘Asian’ group results from an average of deprivation levels for sub-groups that are vastly different, which can be seen by comparing the relatively less deprived ‘Chinese’ subgroup and relatively more deprived ‘Bangladeshi’ subgroup that are averaged into the ‘Asian’ grouping. It also allows disparities associated with minoritized white groups, such as ‘White: Roma’ to be visualised.

For $PM_{2.5}$, Bangladeshi, Pakistani and White Roma groups experienced the largest disparity, with weighted emissions on average 40%, 40% and 36% higher, respectively compared to matched white English, Welsh, Scottish, Northern Irish or British populations. For $NO_x$, the greatest disparities were for Chinese, Arab and Bangladeshi communities who experience weighted emissions 100%, 91%, and 89% higher, respectively than white populations of matched deprivation status. Disparities were observed for minoritized ethnic groups regardless of deprivations status, for example Chinese and Indian communities are, as a whole, less deprived than the England all-population average, but experience higher emissions than the white populations with matched IMD score. Road transport, domestic combustion and industry were the three sectors with the largest contributions to the observed disparity for both $NO_x$ and $PM_{2.5}$.

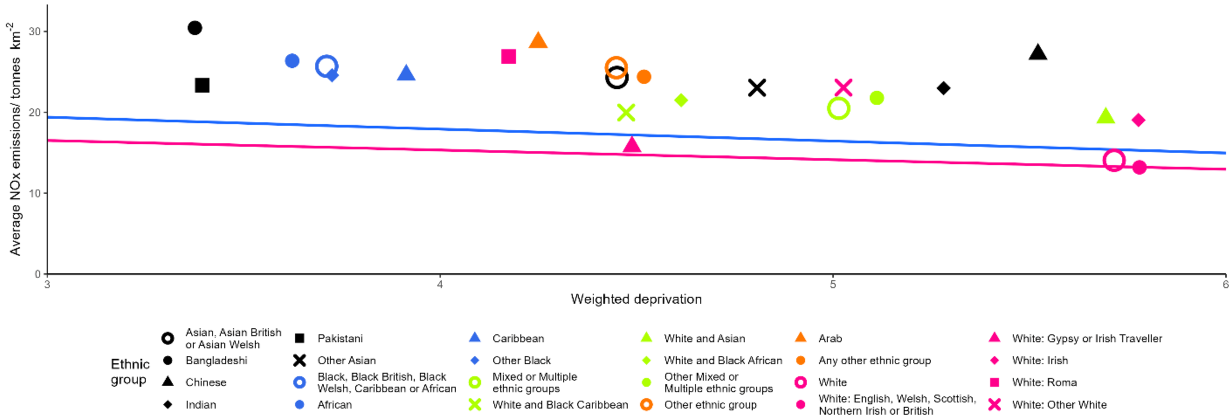


*Figure 4.3. Population weighted $NO_x$ emissions for ethnic groups defined, in the 2021 census, with mean IMD classification based on 2019 data and emissions data taken from the NAEI for 2019. The blue line represents the deprivation-based inequality present in the population as a whole. The pink line represents the deprivation-based inequality present for the majority white ethnic group. From Gray et al., 2024.*

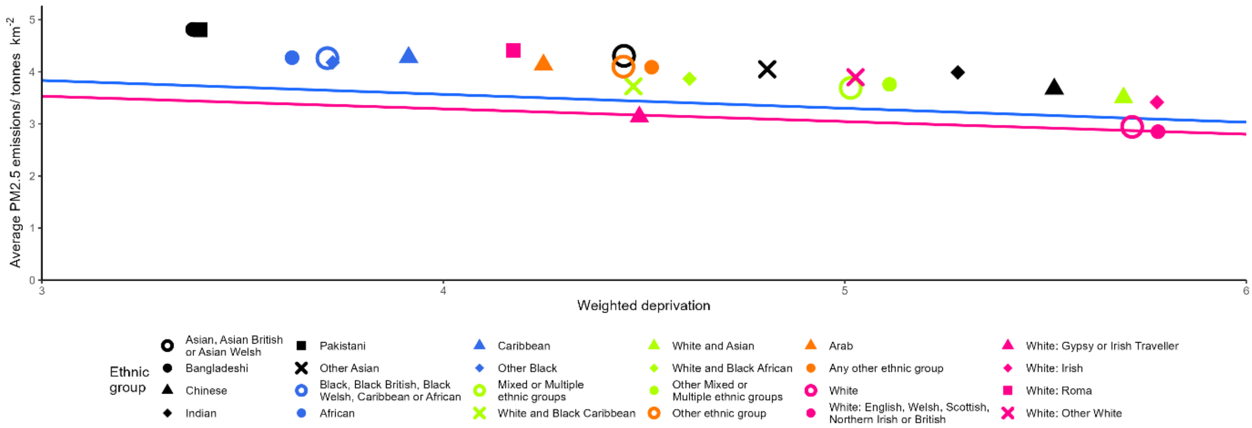


*Figure 4.4. Population weighted primary $PM_{2.5}$ emissions for ethnic groups defined, in the 2021 census, with mean IMD classification based on 2019 data and emissions data taken from the NAEI for 2019. The blue line represents the deprivation-based inequality present in the population as a whole. The pink line represents the deprivation-based inequality present for the majority white ethnic group. Source: Gray et al., 2024.*

The influence of place is also partially explored. Using Rural Urban Classifications it was observed that in urban and town settings minoritised ethnic groups experience higher emissions of both $NO_x$ and $PM_{2.5}$ compared to majority white counterparts living in the same geographies and at the same level of social deprivation. This disparity was not observed for more rural locations. It is also shown that while deprived LSOAs with higher fractional populations of minoritized ethnic groups are concentrated in urban areas, deprived LSOAs with high fraction majority white populations are also visible in coastal locations, notably on the North East and West coasts where emissions are lower.

According to Woodward et al. (2024) road transport and non-industrial combustion (mainly due to domestic wood burning) were identified as the two UK sectors contributing the most to

$PM_{2.5}$ exposure differentials in England when evaluated using the Index of Multiple Deprivation (IMD), both having greater emissions contributions in more deprived areas. Whilst it was recognised that there may be model bias in the evaluation of non-industrial combustion leading to an overestimation of the concentrations in deprived areas and an underestimation of concentrations in wealthy areas, it is clear that urban sources of primary $PM_{2.5}$ contribute disproportionately to geographical differentials in exposure in England due to the greater proportion of poorer households in towns and cities as compared to rural areas.

### Industrial pollution

Walker et al. (2007), investigated the local impacts of industrial pollution, and who in society experienced these impacts through living near to emissions sources. The paper reported on an Environment Agency study, examining the distribution of sites coming within the Industrial Pollution Control (IPC) regime against patterns of deprivation. The analysis provided evidence that IPC sites in England were disproportionately located and clustered together in deprived areas and near to deprived populations. In discussion, Walker et al. identified the methodological limitations of the analysis, including the differences between proximity, risk of harm and the difficult policy questions arising from environmental inequality.

## 4.2. Contributions to emissions and environmental justice

Emissions data has also been used to evaluate who in society is producing the air pollution emissions that lead to disparities in the pollutant concentrations experienced. Mitchell and Dorling (2003) found that the relationship between poverty and $NO_2$ concentrations was not a simple linear relationship, as while the most deprived tend to experience the highest ambient $NO_2$ concentrations, the most affluent also experienced higher than average concentrations. In terms of emissions, they considered initially car ownership since road transport was the dominant source of $NO_2$ emissions. They found that the areas with the lowest rate of car ownership experienced on average the highest $NO_2$ concentrations. When considering $NO_x$ emission and concentrations against poverty together (Figure 4.5), it was proposed that those wards that had the lowest local community emissions of $NO_x$ (based on household private vehicle emissions) and which experienced the greatest $NO_2$ concentrations, were also the most deprived demonstrating a significant environmental injustice.

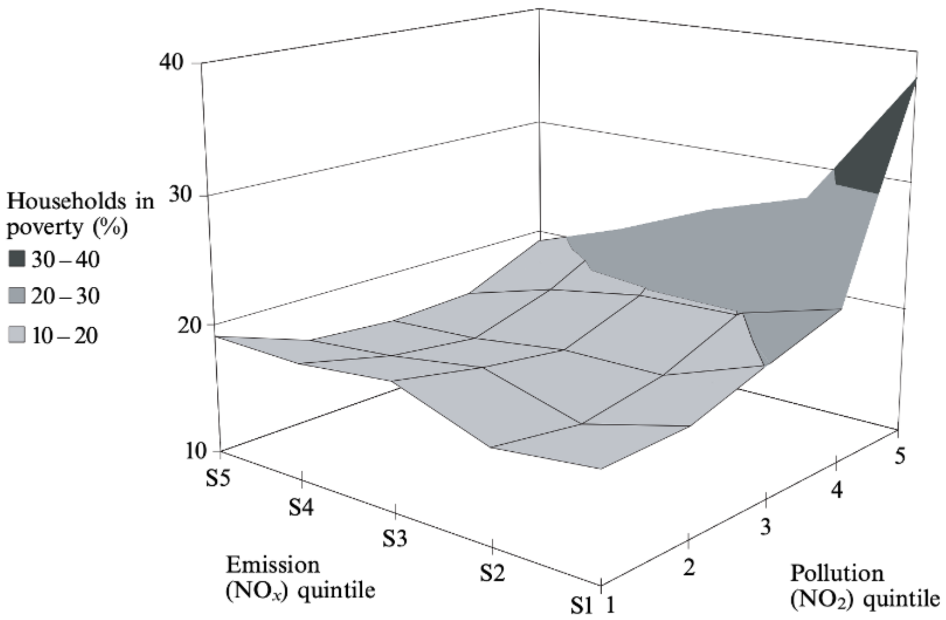


*Figure 4.5. Poverty rate by $NO_x$ emission and ambient $NO_2$ air quality for 10,444 British wards in 1999. S1 is the quintile with lowest local community emissions of $NO_x$, and S5 the highest. Quintile 1 have the lowest concentrations of $NO_2$ and quintile 5 the highest. Source: Mitchell and Dorling, 2003.*

An update to this analysis was carried out by Barnes et al. (2019) estimating that LSOAs with a high proportion of young adults have $NO_2$ concentrations that are twice as high as the average and $NO_x$ emissions that are almost five times higher. Barnes et al. then used vehicle MOT data to show, like Mitchell and Dorling, that those who experience the highest $NO_2$ concentrations were responsible for the lowest emissions from private vehicles (Figure 4.6). The census-based socioeconomic characteristics identified that private vehicle emissions in predominantly higher socioeconomic areas had the highest vehicle $NO_x$ and PM emissions per household, through having higher vehicle ownership, owning more diesel vehicles and driving further.

The implications of the data are that the elevated concentrations experienced by households in poverty are the result of emissions generated by people living in other, generally more affluent, areas. This research concludes that despite a decade of efforts to reduce air pollution, significant inequalities in exposure exist and the same issues of environmental injustice persist.

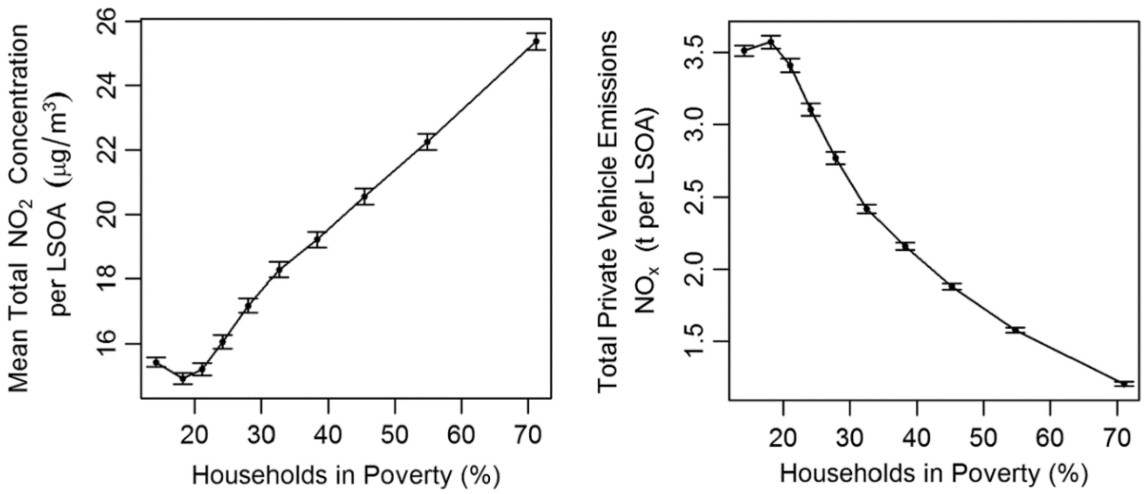


*Figure 4.6. Percentage households in poverty against annual mean $NO_2$ concentrations (left) and total private vehicle $NO_x$ emissions (right). Error bars indicate 95% confidence intervals (CIs).*

# Chapter 5 – Differentials in the distribution of ambient concentrations

Chapter 4 provides evidence from literature that significant insight can be gained from examining differentials in how air pollution emissions are distributed, and that emissions control is ultimately the most direct policy intervention. However, it is differences in atmospheric concentrations that ultimately drive different health outcomes. Chapter 3 highlighted that geographic and meteorological factors play an important part in different parts of the UK and this leads to individuals experiencing different concentrations of ambient pollution because of those factors. However, these external factors explain only some of the differences experienced and in this chapter the differentials that exist in ambient concentrations as a function of other social, economic and demographic factors are considered. Whilst the UK is a leading producer of academic research on air pollution and atmospheric chemistry, the literature on differentials in concentrations in a UK context is rather limited. Hence, the first section of this chapter considers international studies, but it is important to recognise that findings from one country may well not translate directly to the UK.

## 5.1 International evidence

A review by Hajat et al. (2015) provided a global view of ambient criteria air pollutants ($O_3$, $PM_{10}$, $PM_{2.5}$, CO, $NO_2$, $NO_X$ and $SO_2$) and potential inequalities related to socioeconomic status. Across 37 studies, the authors concluded that most communities of low socioeconomic status in North America experienced higher ambient concentrations of air pollution, while in Europe, results were more mixed, and did not provide evidence either way. In Asia, Africa, and other parts of the world the number of papers was small but showed similar results to that of North America.

Another review (Fairburn *et al.*, 2019), investigated social inequalities in exposure to ambient air pollution and found that higher deprivation indices and low economic position are usually linked with higher levels of ambient pollutants $PM_{2.5}$ and $PM_{10}$ and $NO_2$ in the WHO European Region. The review found that there was also evidence that minoritized ethnicity communities experienced variable exposure in comparison to the majority population (Fairburn et al. 2019).

German et al. (2018) considered air pollution, noise and social deprivation across Europe, comparing predicted concentrations of $NO_2$, $PM_{10}$, $PM_{2.5}$ and $O_3$, averaged across 'Nomenclature of Territorial Units for Statistics' (NUTS) level 2 and 3 regions and Urban Audit Cities[1] with a range of different social indicators. Exposure to $NO_2$ tended to be higher in less deprived regions for most indicators; the association was strongest for economic measures of deprivation (GDP per capita and household income). In contrast, for $PM_{10}$, $PM_{2.5}$ and $O_3$ pollution exposure tended to be higher in more deprived areas. The strongest associations between economic deprivation and pollution exposure were seen for $PM_{10}$, with regions both relatively deprived and polluted occurring in eastern and southeastern parts of

[1] Different countries have different classifications, but for comparison, English NUTS2 regions were counties, while NUTS3 regions were districts or groups of unitary authorities.

Europe. For $O_3$ exposure, the strongest associations were found with long-term unemployment and higher-education deprivation, with regions that were both relatively deprived and polluted situated in southern parts of Europe. The authors noted that there was significant variation in correlations depending on the geographic resolution of the analysis, with these Europe-wide conclusions potentially a function of the coarse resolution of the data; for example, a NUTS3 area might be low deprivation on average but still contain areas of high deprivation.

In an international review of more spatially-refined, city-specific distributional analyses, Young et al. (2023) observed that primary pollutant concentrations are typically highest in the centre of cities and along main traffic routes. They noted that how this pattern relates to possible inequity depends on the extent to which these central areas are home to wealthy or deprived communities, with different cities having grown in different ways. For example, Cesaroni et al. (2012) observed that the centre of Rome, which will benefit most from policies to reduce traffic emissions, is disproportionately home to the least deprived communities. Padilla et al. (2014) showed that while Lille, Marseille and Lyon have more deprivation in their centres where air pollution concentrations are higher, the opposite was the case in Paris. Young et al. (2023) thus concluded that exposure for different groups is highly specific to each city, being influenced by historic trends of where jobs and lower-cost housing are found.

## 5.2 UK air pollution concentrations, and links to socioeconomic status, ethnicity and age

Several studies, outlined below, have explored air pollution inequalities at national, regional and city levels. The studies vary in focus and can therefore sometimes be difficult to interpret as a collective evidence base (although a series of reports is available for London). However, there are some clear conclusions which can be drawn, particularly that the most deprived urban areas correlate with the areas of high air pollutant concentrations, and whilst there has been action to successfully reduce air pollutant concentrations, differentials remain and are predicted to remain unless further targeted action is taken. In this section AQEG review studies specifically undertaken in England, Wales and Scotland, but note that none are available for Northern Ireland.

**England**

Walker et al*.* (2005) found a very strong social gradient for nitrogen oxides and $PM_{10}$ in England with the most deprived areas experiencing the worst air quality; including those most likely to be living in areas exceeding the air quality standards. The analysis of Fecht et al. (2015) found that inequalities were largely an urban phenomena and that ethnically diverse neighbourhoods had the highest air pollution concentrations, but with associations that varied between England and the Netherlands. Higher concentrations were found in the most deprived 20% of neighbourhoods in England, and that concentrations in both countries were higher in neighbourhoods with >20% non-White, after adjustment for urbanisation and other variables, such as area level deprivation. They also found that associations between air pollution concentrations and socioeconomic characteristics, ethnicity and age were found to be complex and varied by country, by urban or rural setting and by subpopulation. In addition, whether a neighbourhood is urban or not was found to be one of the strongest determinants of environmental inequality and since $PM_{10}$ and $NO_2$ are markers for traffic-related pollution, measures to reduce air pollution should focus on city transport. They also reported that whilst air pollution concentrations were generally higher in deprived areas, the associations found were often complex, especially between sensitive population subgroups.

Across Britain from 2001–2011, Mitchell et al. (2015) found that the most deprived areas bore a disproportionate and rising share of poor air quality including non-compliance with air quality standards, despite air quality improvements being greatest in the least deprived areas.

Jephcote and Chen (2012) suggested a link between ethnicity and deprivation combined to have a negative impact on some ethnic groups in Leicester for $PM_{10}$ and linked it directly to health outcomes. The study found that Afro-Caribbean and certain South Asian groups are significantly and positively associated with respiratory based hospital admission whereas proportion of Indian residents within an area to be significantly and negatively related to risk of respiratory based hospital admissions.

Urban-rural differentials in exposure to air pollution and mortality burden in England were estimated by Milojevic et al*.* (2017). A non-linear relationship was observed between pollution concentrations and socioeconomic deprivation which varied by urban-rural status. Milojevic et al. (2017) noted that higher ozone concentrations were usually found in less deprived areas.

**Wales**

Brunt et al. (2017) found that in Wales, the most deprived areas are exposed to the highest levels of $NO_2$, $PM_{10}$, and $PM_{2.5}$. Horton et al. (2023) used 1 $km^2$ average modelled concentrations of $NO_2$ and $PM_{2.5}$ to generate population-weighted averages at LSOA level across Wales for the period 2012-2018. These were compared against the Welsh Index of Multiple Deprivation. Over this period, annual mean concentrations of both pollutants declined, benefiting all deprivation groups, but clear discrepancies remained between the least and most deprived areas in all years (Figure 5.1). The authors also reported that a higher proportion of people aged >65 live in areas of Wales with lower $NO_2$ and $PM_{2.5}$ when compared with younger people.

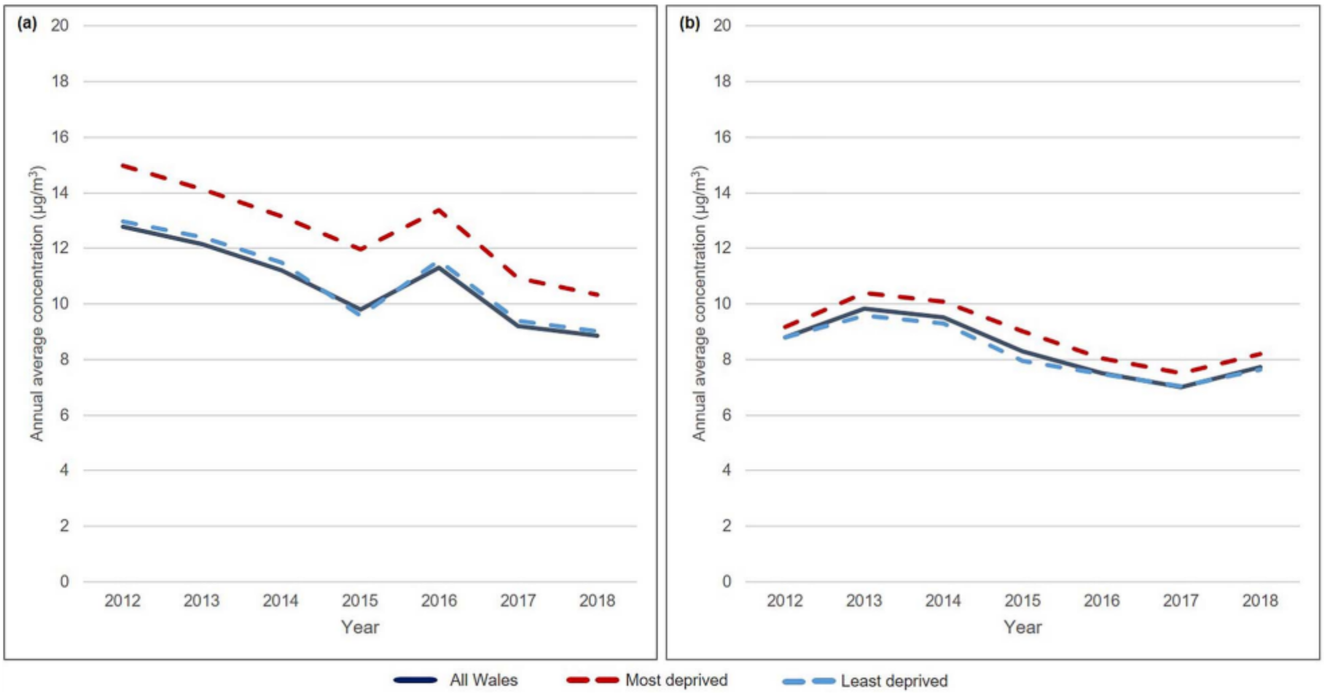


*Figure 5.1. Annual average concentration of (a) $NO_2$ and (b) $PM_{2.5}$ air pollution — all Wales and most (red dots) / least deprived (blue dots) areas (from Horton et al. 2023)*

**Scotland**

Fairburn et al. (2005) found a strong pattern of deprivation and poor air quality for all of Scotland.

Bailey et al. (2018) used 1 $km^2$ average modelled concentrations of annual mean $PM_{2.5}$ across Scotland to calculate average concentrations by ‘data zone’ (which equates to a population of ca. 500 to 1,000 people). These averages were compared with the Scottish Index of Multiple Deprivation. Over Scotland as a whole, air pollution concentrations were highest for the most deprived deciles. As deprivation reduced, concentrations also reduced, but reached a minimum around the third to fifth deciles. The two least deprived deciles were characterised by higher concentrations, although remained lower than those in the most deprived areas. For individual cities, the patterns were often quite different, with Aberdeen showing a clear trend for higher concentrations in the most deprived deciles while Edinburgh showed no such pattern (Figure 5.2). The authors highlighted the potential for geographical selection bias, and the importance of spatial granularity when considering distributional inequality.

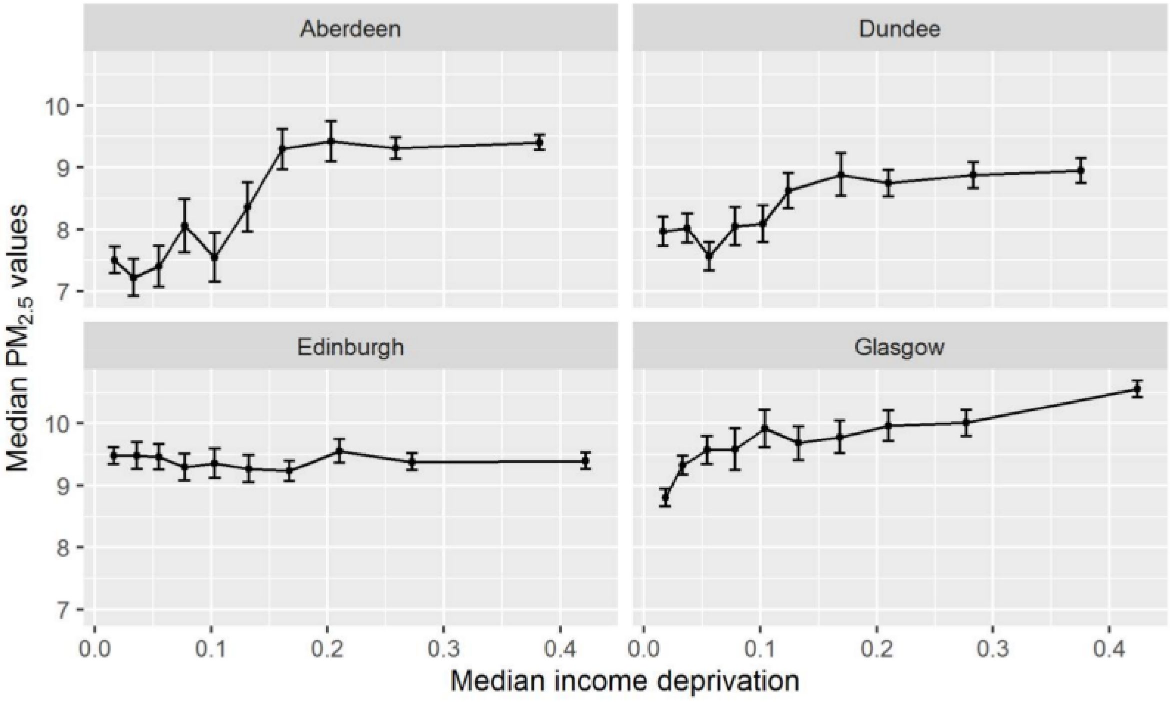


*Figure 5.2. Median $PM_{2.5}$ vs Median Income Deprivation in 2004 in four Scottish Cities, from Bailey et al. (2018). Deprivation deciles and associated median scores defined at national scale to facilitate comparison.*

**London**

Tonne et al. (2018) quantified the socioeconomic and ethnic inequalities of air pollution exposure outside at place of residence and also to personal exposure, assuming this to be a better estimate of 'true' exposure. The two exposure metrics were used as variables in relationships between household income, area-level income deprivation and ethnicity, using quantile and logistic regression.

They observed inverse patterns in disparities in air pollution when estimated at residence versus personal exposure with respect to household income. Specifically, compared to the lowest income group, the highest income group had lower exposure to ambient $NO_2$ at place of residence but higher personal $NO_2$ exposure, which was driven largely by their transport exposure. Although there was significant uncertainty in predicting personal exposure, the authors concluded that socioeconomic disparities in overall 'total' air pollution exposure were different for that estimated based on residential address, and that this had important implications for environmental justice and confounding in epidemiology studies.

One of the most comprehensive studies, particularly in terms of duration, has been a series of studies commissioned by the Greater London Authority (King et al., 2013; King et al., 2017; King et al., 2019; Williamson et al., 2021, Brook et al., 2023, King et al., 2023). These considered inequalities in exposure to $NO_2$, $PM_{10}$ and $PM_{2.5}$ in London, including how this had changed, and might be expected to continue to change. These studies used OA and LSOA granularity and have consistently shown that *"the most deprived communities of London still more commonly live in the most polluted areas, however concentrations have declined faster in areas of deprivation and more markedly since 2016."* The reports show that this pattern is independent of differences between inner and outer London.

Average concentrations have fallen appreciably since 2013 and are expected to continue to fall in the near future, and areas with the highest historical concentrations have seen the largest improvements. This means that, in terms of absolute concentrations, the gap between the least and most deprived areas reduced between 2013 and 2019 (cf. Mitchell et al., 2015). However, despite these historical improvements, Brook et al. (2023) note that *"… unless further significant action is taken, the differential of pollution experienced between the least and most deprived will remain"* and that *"Further policy developments, such as the expansion of the ULEZ to be London-wide may lead to greater reductions in air pollution and reduce inequalities in exposure"*. Similarly, in the projections to 2030, while all deprivation groups are expected to see improvements which are much larger than concurrent between-group disparities, disparities across different groups are predicted to remain (Figure 5.3).

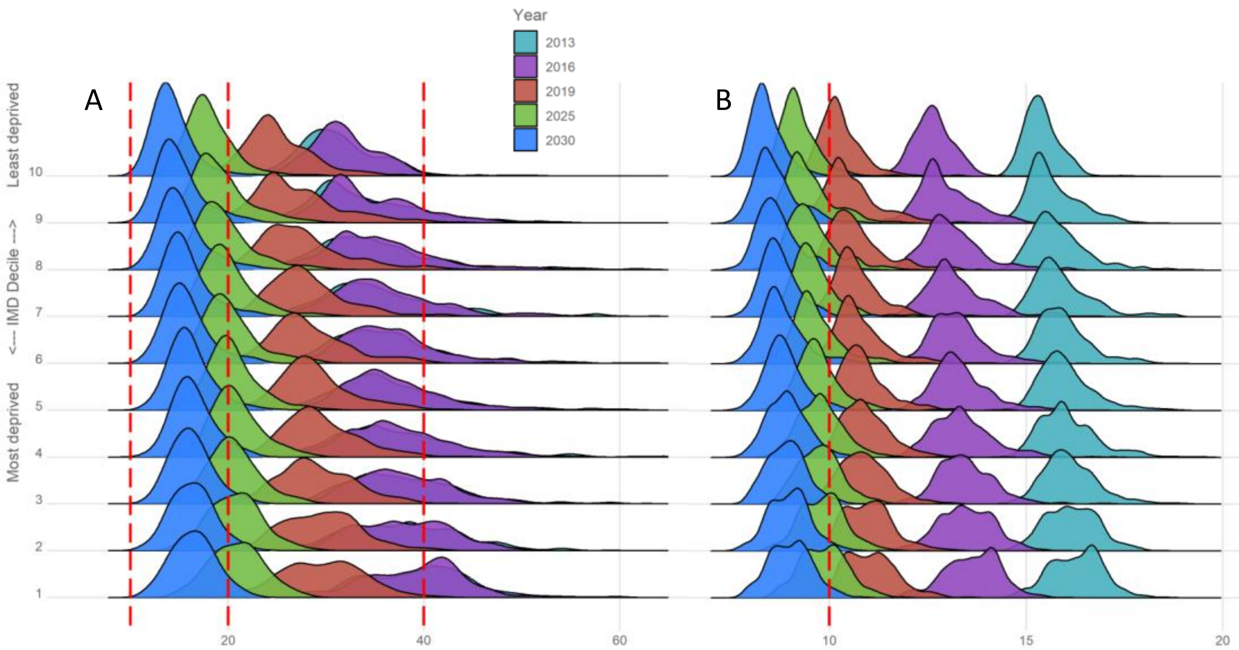


*Figure 5.3. Average concentration exposure distributions within IMD Decile across years. A) $NO_2$, B) $PM_{2.5}$. Red dotted lines show a range of alternative assessment metrics (from Brook et al., 2023).*

Brook et al. (2023) also showed that the areas in London with the lowest $NO_2$ and $PM_{2.5}$ concentrations had a disproportionately white population, and that this did not change significantly between 2013 and 2019. The difference was more pronounced in outer London than inner London. In outer London, the lowest concentration decile in terms of $NO_2$ was 71% white, compared with 56% white in Inner London. White and Asian populations were underrepresented in the most polluted areas in comparison to the general population, whereas Black, Mixed Multiple and Other populations were overrepresented, and there was little discernible change observed over time. However, they showed that half of areas with higher concentrations generally had a representative spread of the population by ethnicity. Diaspora groups with the highest levels of exposure to $NO_2$ and $PM_{2.5}$ were North American, Middle Eastern and Eastern Asian, with this distribution changing little since 2013.

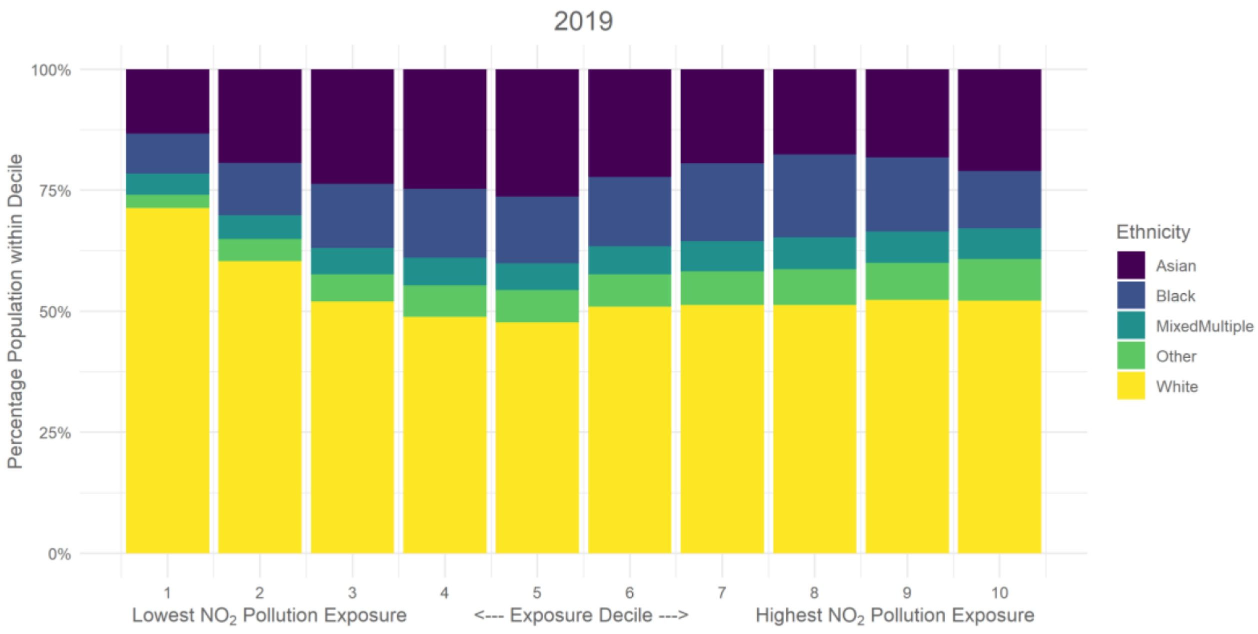


*Figure 5.4. Exposure to $NO_2$ concentration deciles in London for 2019 resolved by broad ethnicity characteristics. From Brook et al. (2023).*

Brook et al. (2023) included an assessment of air pollutant concentrations at vulnerable receptor sites (schools, hospitals, and care homes). They concluded that:

- Progress has been made in recent years, and was projected to continue, with the number of vulnerable receptors exposed to concentrations above the WHO interim target annual mean concentrations predicted to reduce significantly by 2030. For $NO_2$, in 2013 and also in 2016, 100% of schools, hospitals and care homes were exposed to concentrations above the WHO interim guideline of 20 µg $m^{-3}$. This was forecast to reduce to 7%, 28% and 2%, respectively, in 2030. For $PM_{2.5}$ the percentage of schools, hospitals and care homes exposed to concentrations above the WHO interim target of 10 µg $m^{-3}$ was forecast to reduce from 100% for all to 5%, 20% and 1%, respectively, over the same period.

- Schools in inner London were on average exposed to higher concentration of air pollution compared with schools in outer London and this remained consistent over time, however only 51 (4% Inner London) schools in Inner London remained in exceedance of the legal limit for $NO_2$ in 2019 and this was forecast to be none in 2025.

There was a weak but positive correlation in 2019 between the percentage of pupils eligible for free school meals and the concentrations of air pollution that a school is exposed to (excluding private schools where no pupils are eligible for free school meals). This relationship was forecast to weaken over time as the range of air pollutant concentrations that schools are exposed to decreases.

## 5.3 Differentials in concentrations arising from traffic sources

The historically large emissions contribution from road transport to urban air pollution concentrations, and specifically to ambient $NO_2$ has led to many studies specifically evaluating differentials and association of transport sources. It is important to note the rapidly changing nature of transport exhaust emissions and so literature and associations found can potentially go out of date relatively quickly.

Mitchell and Dorling (2003) provided the first national study of the environmental injustice of air quality in Britain, focusing on ward-level exposure to $NO_2$ concentrations, a pollutant which at the time was very strongly associated with vehicles, and comparing these estimates to the local contribution to vehicle emissions. The analysis presented evidence of environmental injustice, concluding that, '*those communities that are most polluted and which also emit the least vehicle pollution tend to be amongst the poorest in Britain*'.

Barnes et al. (2019) also assessed the UK environmental injustice issue of exposure to road traffic related air pollution vs. the emissions generated, once again focusing on $NO_2$. The study identified discrepancies between traffic-related emissions generation and exposure by socioeconomic and demographic groups, demonstrating environmental and social injustice. The analysis for England and Wales updated Mitchell and Dorling's work, using 2011 UK Government pollution and emissions data with 2011 UK Census socioeconomic and demographic data, demonstrating that areas with the highest proportions of under-fives, young adults and poorer households, had the highest concentrations of traffic-related pollution.

Brook et al. (2023) is a recent report on the issue of transport emissions and links to socioeconomic and other factors, part of a series of studies of London for the GLA. The study included an assessment of the main traffic routes in London (called "red routes") and concluded that:

- Communities living along red routes were exposed to higher air pollutant concentrations, though these concentrations were expected to reduce over time. The red route population exposed to $NO_2$ concentrations exceeding the WHO interim guideline is forecast to reduce from 100% (1.1 million) in 2016 to 76% (1 million) in 2025 and to 19% (0.2 million) in 2030. As red routes are designed for main traffic flows, they are likely to be the last places in London to meet the WHO interim targets unless further action is taken.
- 95% of the population exposure to $NO_2$ concentrations above the WHO interim target in 2030 were in Inner London, which represented over a third of the inner London red route population compared with 10% of the inner London non-red route population.
- Almost the entire (99%) population exposed to concentrations above the $PM_{2.5}$ WHO interim guideline in 2030 are in inner London or along the North and South Circular. This represented over a third of the 2030 Inner London red route population, compared to 5% of the Inner London non-red route population.

Figure 5.5 shows the annual mean concentrations at Output Area (OA) Level for $NO_2$ by red route group for the different years of the London study.

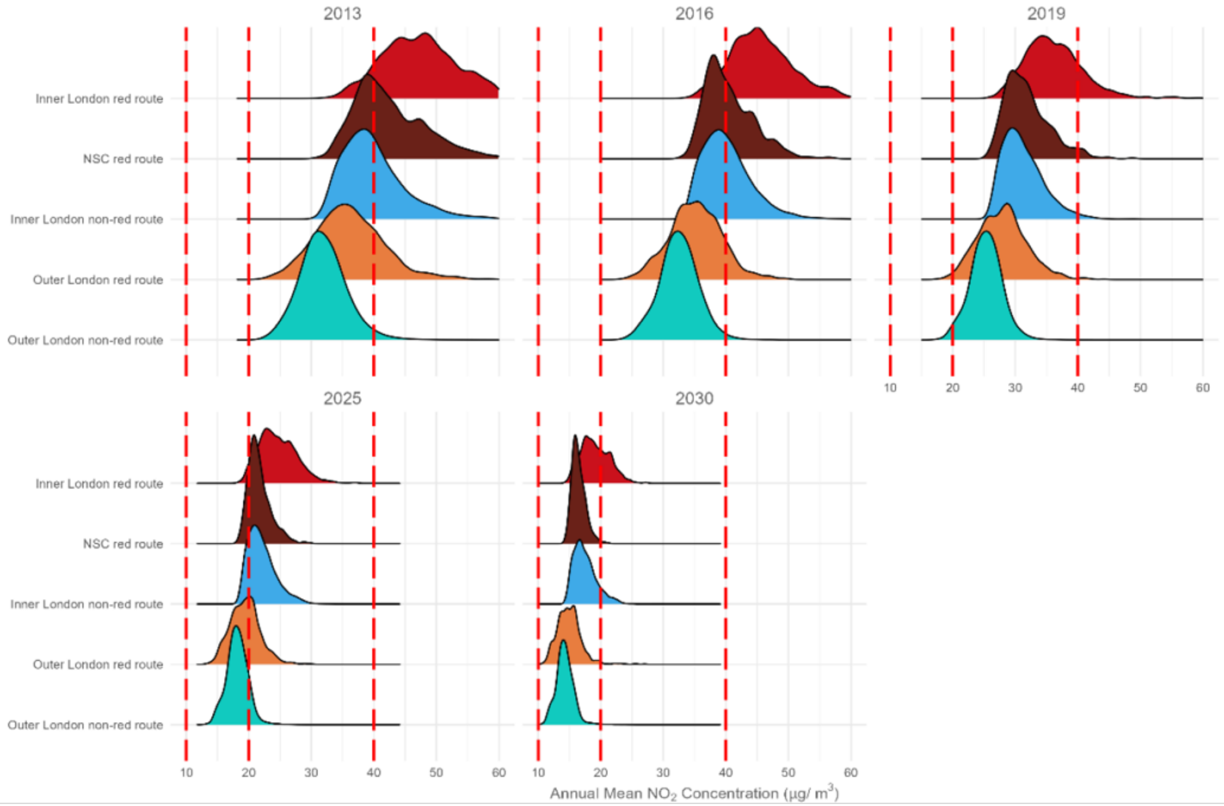


*Figure 5.5. Annual mean concentrations at Output Area (OA) Level for $NO_2$ by London 'red route' group, 2013, 2016, 2019, 2025 and 2030. The dotted red lines represent (left to right) the $NO_2$ WHO guideline, WHO interim target and England and Wales limit value. NSC refers to North and South Circular red route populations.*

## 5.4 Conflicting evidence

The paper of Hajat et al. (2015) summarised the results of European studies of air pollution and inequalities as providing 'mixed' evidence of the poorest in society being exposed to the highest air pollution. It is not surprising therefore that some remain to be convinced of the effect, or at least the universality of the association. Briggs et al. (2008) is a case in point stating that previous studies of environmental inequity have indicated considerable complexity in the associations involved, which merit further investigation. The Briggs et al. study investigated how environmental inequity in England varied by: (a) different environmental pollutants; (b) different aspects of socioeconomic status; and (c) different geographical scales and contexts (urban vs. rural). Associations were quantified between the Index of Multiple Deprivation (IMD2004) and five sets of environmental pollutants (relating to road traffic and industry, but also others such as electro-magnetic radiation, disinfection by-products in drinking water and radon), measured in terms of proximity, emission intensity and environmental concentration.

Briggs et al. found that associations were generally weak ($R^2 < 0.10$), with the strongest associations occurring between crime, living environment and health, rather than causative factors of inequity, such as income, employment or education. Associations were also reported to become stronger with increasing levels of spatial aggregation. Briggs et al. finally concluded that although "stronger associations tend to be found with measures of air pollution than other types of hazard, and with environmental concentrations rather than proximity to source or emissions", the results suggested that health outcomes arising from environmental inequities, were likely to be limited.

## 5.5 Implications for confounding in epidemiological studies

As was noted by Tonne et al. (2018) and Briggs et al. (2008) in epidemiological research, there are difficulties in addressing the influence of socioeconomic confounding, which is essential to obtaining robust associations with health outcomes. In London, Goodman et al. (2011) compared exposure to $NO_X$ with socioeconomic markers and scales of measurement, focusing on traffic-related air pollution. This research compared traffic-related air pollution with area- and individual-level socioeconomic position (SEP), concluding that 'mean air pollution concentrations were generally higher in postcodes of low SEP as classified by small-area markers of deprivation (Index of Multiple Deprivation (IMD) domains) and by A Classification of Residual Neighbourhoods (ACORN) geodemographic marker.

However, the complexity of these associations was demonstrated by this relationship being reversed in central London and for SEP markers for education. With regards to confounding issues, Goodman et al. (2011) concluded that area-level adjustment for socioeconomic confounding using both ACORN and IMD was suitable for epidemiological studies.

# Chapter 6 - Differentials in air pollution related to indoor environments

## 6.1 Overview and international context

Air quality in the indoor environment has received an increasing amount of attention in recent years. Indoor air pollution is now becoming one of the most important sources of exposure for many individuals as outdoor pollution concentrations reduce and air exchange rates decrease in modern buildings. This has been accompanied by advances in understanding of sources and processes, as new instrumentation and observations have been brought to bear. This has been the subject of the previous 2022 AQEG report *Indoor Air Quality*.

Indoor air pollution also has many intrinsic aspects that have relationships with societal inequalities, thus can increase the health burden on disadvantaged groups. Internationally, the World Health Organization (WHO, 2023) has listed this as a priority area for action, stating "*Significant policy changes are needed to rapidly increase the number of people with access to clean fuels and technologies by 2030 to address health inequities, achieve the 2030 Agenda for Sustainable Development, and mitigate climate change*," and, "*Women and children disproportionately bear the greatest health burden from polluting fuels and technologies in homes as they typically labour over household chores such as cooking and collecting firewood and spend more time exposed to harmful smoke from polluting stoves and fuels*."

In the USA, the Environmental Protection Agency has made Indoor Air Quality a priority area as part of its Environmental Justice agenda, which includes community engagement and trying to ensure homes conform to indoor air quality standards among disadvantaged groups.

Exposure to indoor pollution can also be magnified by increased time spent indoors. This can be associated with higher unemployment rates (Krueger and Mueller, 2012) or a general tendency to engage in indoor recreational activities associated with lower socioeconomic status (Tandon et al., 2012).  However, since the COVID-19 pandemic, a sustained higher proportion of individuals have identified as working at home partly or fully, with the higher income groups more likely to do so (ONS, 2023). This recent trend may mean that higher income individuals may be spending more time at home, which may have poorer air quality than a conventional office (chi is more likely to be air conditioned and have air fliters). On the other hand, this may lead to lower exposure to air pollution during commuting. Finally, those of a lower socioeconomic status are also more likely to have pre-existing health conditions such as asthma, which may make them more sensitive to poor indoor air quality.

The indoor air pollutants of interest can broadly be described as either 'conventional' air pollutants, which are analogous to if not identical to outdoor air pollutants such as $NO_2$, $PM_{2.5}$, VOCs and CO, or 'biological', which are produced by biological sources and are more specific to the indoor environment, and this section is divided accordingly. This report does not consider radon, although it is worth noting that in the USA, this is considered a major facet of inequality-driven exposure through properties located in areas with strong geological sources of radon with inadequate ventilation (EPA, 2024). However there is evidence that radon concentrations are actually higher in more affluent properties, including in the UK, which can be due to the size of the property and draughtproofing (Kendall et al., 2016; Ferguson et al., 2020).  This chapter focuses mainly on exposure in residential properties, however it should be noted that lower socioeconomic groups may also experience higher workplace indoor air pollution exposure, through working in buildings that are less well maintained or located in less desirable areas, or through performing jobs with generally higher occupational exposures (Fujishiro et al., 2010).

Research publications on indoor air pollution are becoming more common (e.g. Farmer et al., 2019) and at the time of writing a number of UK-based research activities are being directed to this issue as part of the NERC Clean Air Programme (https://www.ukcleanair.org/), which includes (but is not limited to) projects such as INGENIOUS, WellHome and IAQ-EMS. But there are fewer published studies quantitatively linking differentials in air pollution specifically to factors such as deprivation, or demographics. One example however is Ferguson et al. (2021), who performed a comprehensive evaluation of mechanisms with relevance to London specifically. While stopping short of a full quantitative assessment, it proposed a systems framework, shown in Figure 6.1, which demonstrates how various socioeconomic and domestic environmental health issues can link to one another. Likewise, mechanisms by which PM exposure specifically can be influenced by socioeconomic factors in the USA were also identified in a recent report on indoor PM by the National Academies of Sciences, Engineering, and Medicine (2024), but this too did not provide a full assessment.


*Figure 6.1. Systems framework for inequalities and indoor air pollution relevant to London proposed by Ferguson et al. (2021), adapted using [https://kumu.io/jonathontaylor/indoor-air-pollution#systemic-inequalities](https://kumu.io/jonathontaylor/indoor-air-pollution#systemic-inequalities).*

Generally, while the individual mechanisms linking societal and demographic factors and indoor air quality have been identified and systems-based approaches proposed, the lack of comprehensive quantitative assessments that would aid targeted interventions represents a current gap in the research, although addressing this presents many challenges including regulatory, policy and behavioural. The exception to this is the issue of bioaerosol exposure relating to damp and mould (see 6.3), where quantitative assessments have been published that supplement the evidence concerning the wider issues concerning housing quality.

There are potentially differential impacts on indoor air quality that are associated with decarbonisation of homes. Some decarbonisation actions such as electrification remove pollution sources from inside homes. However, the effects of increasingly energy-efficient homes on damp and mould, and the accumulation of primary pollutants such as $NO_2$, $PM_{2.5}$ and volatile organic compounds across different types of housing in the UK is uncertain.

More systematic evaluation of air quality in different building types in the UK is needed, including the impacts of regulations guiding construction of energy-efficient new buildings and those that are representative of on homes and businesses undergoing energy efficiency retrofits. Noting that comprehensive coverage of observed air quality in homes is unlikely to be achieved, better predictive capability of exposures is needed to inform health research and policymaking. This in turn must be informed by a thorough understanding of the relevant sources and processes influencing exposure.

## 6.2 Conventional air pollutant sources

There are multiple sources of ‘conventional’ indoor pollutants and many of these can be linked to social deprivation. A significant cause of indoor pollution is the infiltration of outdoor air into the indoor environment; if it is known that certain socioeconomic groups live in areas with higher outdoor pollution levels as described elsewhere in this report then this will increase indoor concentrations as well.

There exist several direct air pollution sources within the indoor environment, which are detailed in the AQEG Indoor Air Quality report (AQEG, 2022) and some can exhibit disparities between different socioeconomic groups. Cooking is a major source of indoor air pollution in the form of PM and VOCs and can be a source of disparity in exposure in different demographics (Kashtan et al., 2024). It has been shown that those in lower income groups spend more time cooking on average (Adams and White, 2015), so it follows that this will increase emissions and thus exposure. Gas appliances such as boilers and cookers may contribute more to indoor $NO_2$ and CO concentrations if they are poorly maintained, so these may increase if a resident is unable to afford to properly maintain these, or a landlord is negligent.

The burning of solid fuels for heat can contribute to indoor air pollution in the form of CO, $NO_2$ and PM (Chakraborty et al., 2020). However there is little evidence to suggest that fuel poverty is significantly forcing lower socioeconomic groups to rely on solid fuel burning as a primary source of heat (Ferguson et al., 2021), with increased solid fuel burning occurring among wealthier households as a secondary source of heat to supplement gas or electric heating (Defra, 2020). Thus, this mechanism may represent a tendency for increasing indoor air pollution in response to a higher socioeconomic status. However, the significance of stoves as a source of indoor air pollution may also depend on the quality of installation, operation and maintenance of the stove, which could be expected to be better for wealthier households.

VOCs are also emitted from consumer products such as cleaning and personal care products, DIY solvents, aerosol cans and air fresheners, and these can act as another source of indoor air pollution, along with the by-products of chemical reactions they initiate, in particular formaldehyde, which is thought to have a significant health burden (Clark et al., 2023). There is evidence to suggest that normalised to the number of residents, emissions of

VOCs are greater in higher socioeconomic status households, due to increased consumption of consumer products (Brown et al., 2015).

Smoking represents a particularly hazardous source of indoor air pollution. Asides the direct impacts on the smoker, the indoor environment can be a source of passive exposure, through inhalation of ‘second hand smoke’ but also exposure to residues in household furnishings and dust, so called ‘third hand smoke’ (Wu et al., 2022). Tobacco smoke is enhanced for lower socioeconomic status households in line with increased smoking rates (ONS, 2023), which translates to higher exposures (Ferguson et al., 2020, and references therein). Asides tobacco smoke, there are analogous airborne pollutants present in indoor environments from vaping and types of drug use.

A large driving factor influencing exposure to indoor air pollution in relation to socioeconomic status is the size of the property relative to the number of occupants. With a larger number of rooms, individuals are less likely to need to spend time in proximity to another’s pollution-generating activities, such as smoking or cooking. Also, concentrations are likely to be higher in smaller-volume rooms because of the reduced dispersal from a given activity (Kashtan et al., 2024). Furthermore, strong sources such as tobacco smoke may transfer between apartments in the same building; these factors combine to amplify exposure to all sources of indoor pollution.

Another driver linking indoor air quality and deprivation is the quality of the property and its maintenance. A major factor is ventilation, including opening of windows and mechanical extraction in ‘wet rooms’ such as kitchens and bathrooms. When working well, ventilation can reduce the build-up of pollutants from indoor sources and can also help to prevent damp and mould (see below). While a draughtier property may suffer less from indoor air pollution, too high an air turnover can result in more outdoor pollution infiltrating the building, and other environmental stressors such as cold.

Modern newly built properties normally have engineered ventilation, but older properties may have inadequate or poorly designed ventilation, in particular if they have been retrofitted with draught proof doors and windows for the sake of energy efficiency. It is argued that modern properties may have placed too great an emphasis on energy efficiency at the expense of indoor air quality (McGill et al. 2016). This can represent a mechanism by which a more affluent property could experience higher concentrations than a less well maintained, draughtier property. But conversely, in lower socioeconomic households, features such as vents and mechanical ventilation may be non-functional, either through a lack of maintenance or being deliberately disabled to reduce heating requirements. Higher income groups also have more access to additional technical solutions such as HEPA filtration and dehumidifiers.

## 6.3 Biological pollutant sources

Beyond the ‘conventional’ pollutants, indoor air quality is also heavily impacted by biological sources. Of particular concern are aerosolised spores produced by fungi (‘moulds’) growing on damp materials. These are known to have the potential for detrimental effects on health that include respiratory conditions in young children and causing asthma in sensitised people of all ages (Ingham et al., 2019; Denning and Pfavayi, 2023). The issue of mould in homes was raised to particular prominence in November 2022, when a coroner’s inquest ruled that damp-related mould contributed to the death of Awaab Ishak (aged 2), in social housing in Rochdale, Greater Manchester, in 2020 (HM Judiciary, 2022).

In addition to building surfaces, moulds can infect fabrics and bedding (Woodcock et al., 2006) and high humidities can also increase dust mite populations (Niven et al., 1999), which produce allergens that are recognised as another major cause of asthma. Passive drying of clothes indoors, particularly in smaller properties, can also increase moisture levels (Porteous et al., 2014). Smoking related diseases such as chronic obstructive pulmonary disease (COPD), associated with lower income groups, are exacerbated by indoor mould exposures which can lead to chronic pulmonary aspergillosis CPA (Kosmidis et al. 2023). Under severely squalid conditions, household dust, pet dander, spoiled food and human and animal waste can act as major sources of bioaerosol, VOCs and ammonia (HSE, 2003). Such conditions are associated with households with inadequately cared-for pets and/or very vulnerable individuals, such as those with mental health conditions, the elderly, drug addicts and those with unmet physical disability needs (Snowdon et al., 2007). These air quality problems will compound other health issues associated with the poor hygiene of these conditions.

These issues relating to damp and mould and social inequality are covered in the February 2023 UK Parliamentary Research Briefing *Health inequalities: Cold or damp homes* (Balogun et al. 2023). Referring to the English Housing Survey (DLUHC, 2024)), it reported that an estimated 904,000 homes suffered damp problems, with a tendency towards certain groups occupying these, specifically, “households with an older person living in them, households with a lone parent, households with children, low income households, and households with people from minority ethnic backgrounds”. Further, groups who are most likely to live in homes with damp and mould include people with a long-term illness or disability, and people living in temporary accommodation (DLUHC, 2023).

Damp problems were more common in the private rented sector (11%) compared against social-renting (4%) and owner-occupied (2%) households. However as noted in the report, reliable statistics on these issues can be difficult, as there is a tendency for tenants to avoid reporting such issues, for fear of eviction or rent rises, so these figures may be an underestimate. Additionally, people may also be unaware of their rights as tenants and the report identified inconsistent enforcement between different local authorities. While this would indicate an association with lower quality housing stocks, it must also be noted that more modern or retrofitted homes that have been designed around energy efficiency may also suffer increased damp and associated bioaerosol concentrations (Brambilla and Sangiorgio, 2020).

There is broad scientific and policy consensus around the motivations for reducing damp as a housing problem. According to the English Housing survey, rates of homes classed as having a damp problem (penetrating, rising or condensation) have been decreasing, but not uniformly across different socioeconomic groups (Figure 6.2). In particular, there is a consistently strong association between household income and damp and there are also trends within ethnic groups. While non-White groups were more likely to inhabit damp properties compared to white populations in 2008, they are now equivalent, with the exception of Black households, where there still appears to be a higher tendency for damp homes compared to White households and other ethnic groups that shows a possible upward trend in recent years. Note however the 2020 and 2021 surveys were impacted by

COVID restrictions, meaning the amount of data available and thus quality of the statistics was reduced, so the data from these years should be viewed with caution.

Also based on English Housing Survey data, Clark et al. (2023) found an association between ethnicity and income and an elevated incidence of damp and mould attributed asthma and lower respiratory infections in adults (Figure 6.3). The causal mechanisms linking damp and mould to respiratory impacts are still under-investigated and may be exacerbated by exposures to other pollutants of indoor air, discussed above. However, based on this assessment, they estimated that in 2019, damp and mould in houses could be associated with approximately 5,000 new cases of asthma and 8,500 lower respiratory infections in England, representing 2,200 and 600 disability-adjusted life years (DALYs) lost respectively.

There is currently activity at central and local levels to address the problem of damp and mould in homes, including the implementation of the Social Housing (Regulation) Act in July 2023. Within the act, 'Awaab's Law' mandates social housing landlords to address hazards such as dampness and mould within a time-limited period.

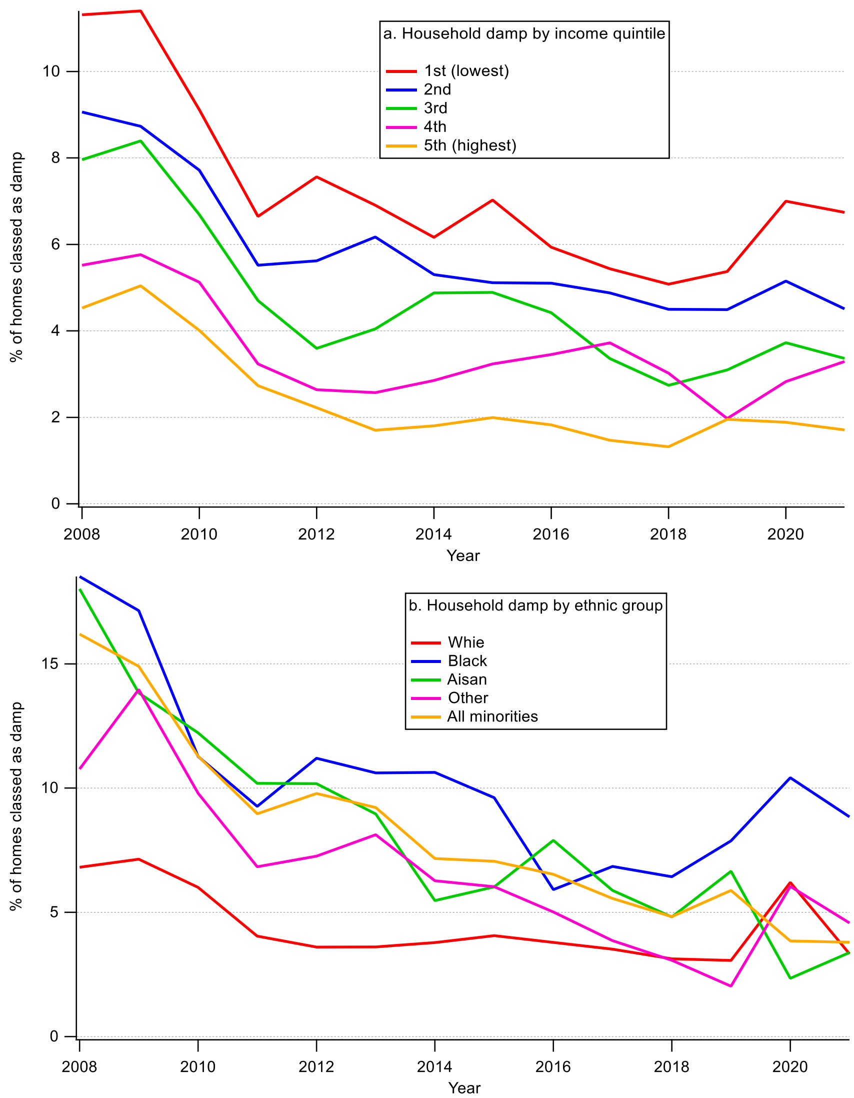


*Figure 6.2. Trends in fraction of domestic properties classed as 'damp' by the annual English Housing Survey, divided by (a) income and (b) ethnicity. Note the surveys in 2020 and 2021 were COVID-impacted and thus resultant data is likely of reduced quality.*

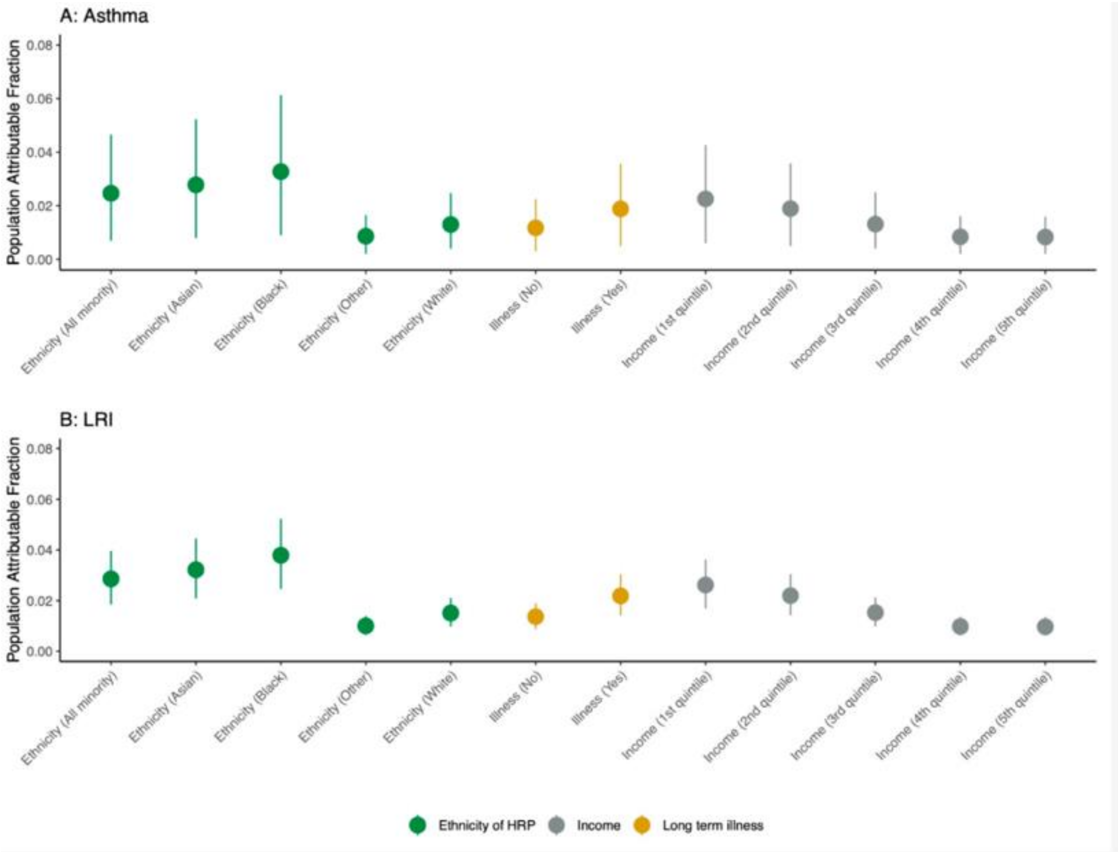


*Figure 6.3. Population Attributable Fractions for asthma (A) and lower respiratory infections (LRI, B) among adults associated with damp and/or mould in English residences 2019. (HRP = Household reference person). Figure from Clark et al. 2023.*

# Chapter 7 – Differentials in air pollution: impact of travel and occupation

## 7.1 Travel related differentials

Most research on differentials in air pollution experienced during the use of transport relates to differences in concentrations or exposure by mode rather than the differences that arise as a function of socioeconomic status or demographic factors, such as age, income, occupation, health-vulnerability or household deprivation.

Travel modes and the consequent exposure to air pollution was the subject of recent reviews by Cepeda et al. (2017) and Mitsakou et al. (2021), encompassing both motorised and active travel. These reviews highlighted that the relatively small proportion of times spent travelling can led to a sizable proportion of daily exposure to some air pollutants. On average people spend around six percent of their time travelling. This has been estimated to contribute up to 30% of their daily air pollutant exposure to some pollutants. Most studies have considered carbon monoxide, black carbon (BC), $NO_2$ and fine and coarse particles. It is acknowledged that patterns for $O_3$ exposure will be notably different.

Pollution concentrations depend on the study setting. For this reason, reviews have not sought to compare a pooled estimate of the concentration or exposure in each mode. Instead, they create concentration and exposure rankings based on the number of studies that conclude that one mode has a greater concentration or exposure compared with another.

Defining exposure as the product of concentration and time, a consensus hierarchy has been found with commuters using motorised modes of transport, (including public transport bus, rail and underground / subway) having greater exposure than those that use active travel modes such as walking or cycling. Within the motorised transport modes, car commuters generally have the greatest exposure, and these are often greater than that experienced by active travellers. For those cities with underground / subway systems, this can be the mode with highest $PM_{10}$ concentrations.  Lowest exposure for rail and active travellers is usually found where railway lines, cycle lanes, and footpaths were located away from roads.

Smith et al. (2016) modelled the exposure of people travelling in London using travel survey data and modelled outdoor $NO_2$ and $PM_{2.5}$. Lower exposures to outdoor air pollution were experienced by those that mainly stayed at home. For those that travelled, use of active modes led to lowest exposure.

Going beyond concentrations and exposure by travel mode, Guzman et al. (2023) compared the potential inhaled dose by transport mode according to individuals' socioeconomic background and travel patterns. The study was conducted in Bogotá, Colombia and considered $PM_{2.5}$, BC and CO. Using micro-environment measurements and a household mobility survey they found differences in exposure that could be explained by travel mode and travel time. They concluded that a significant share of the population in Bogotá is more exposed to air pollution in the transport system, has lower transport accessibility for work and study, and that this share of the population is also poor or very poor. The study pointed to investment in low or zero emission buses as an efficacious intervention especially given the longer bus journey-times for poorer communities that have historically settled at the edges of the city. It follows that faster journey times would also reduce exposure.

Rivas et al. (2017) measured black carbon, particle number and mass concentration on commuting journeys between suburbs and the City of London. Journeys were between 8 km and 12 km, and over 200 trips were measured. Four origin areas were selected to represent the range of average incomes in London and journeys were made by bus, underground and car. Transport mode was found to have a greater influence on mean concentrations and exposure compared with the ambient air pollution in the origin area.  By far the greatest black carbon and PM was measured in the underground and especially the deep cut sections of the network and in carriages with openable windows, though the authors acknowledge that the physical properties of particles in the underground will have led to measurement inaccuracies (this is consistent with later measurement campaigns by Smith et al 2019). Use of the underground showed little difference between the four origin areas studied by Rivas et al (2017), but people in the lower income origin areas made greater use of buses and less use of private cars compared with the more affluent areas. Greater use of buses and the longer journey times by bus led to greatest exposure for the least well off.

Differentials in air pollution concentrations are not confined to commuting for work. Varaden et al. (2021) examined children's exposure to air pollution during their school commute and found that $PM_{2.5}$ concentrations on their morning journey was on average 52% greater than that at school. Beyond that, children who walked along busy main roads were exposed to 33% higher levels of air pollution than those who walked along quieter back streets. Those that travelled to and from school by car experienced greater concentration than those that walked along non-main roads.

## 7.2 Differences in vehicle ownership and access

Knowledge and data collection on traffic systems focuses mainly on flows rather than patterns of the ownership and use of vehicles. It therefore provides only limited insight into the socioeconomic context of the people making these journeys. These might include differentials in availability and access to public transport, vehicle ownership, the types of vehicles owned and the age of vehicles.

Not everyone has access to a car or van. Based on travel surveys, nationally, 22% of households do not have access to a car or van (ONS, 2022). For London this is 36% (London Assembly, 2022). Access to a car has a strong relationship to income. Data from the London Travel Survey for 2019/20 shows that over 80 percent of households with an annual income of less than £5,000 have no access to a car. For those with incomes of greater than £50,000, 10 to 15% of households do not have access to a car and 39% own more than one car. Spatially there are substantial variations within London with over 60% households being car- free in central boroughs, compared with less than 20% in some outer London boroughs (London Assembly, 2022).

Given the differentials in access to a car or van by income it is perhaps no surprise that propensity for active travel (walking and cycling) also varies by socioeconomic status. Analysis of data from the Scottish Household Survey (Olsen et al. 2017) found that the likelihood of an active travel journey was 21% greater in the most deprived quintile of households and that these active travel journeys tended to be longer than those taken by those in wealthier households. Active travel was dominated by walking which comprised 24.9% of all journeys compared with 1.1% cycled. Active travel was more common for urban than rural journeys and for the journeys taken by young people.

Barnes et al. 2019 combined UK census and air pollution data with information from annual vehicle safety (MOT) inspections. Based on data collected around 2011 (e.g. in the period before mainstream adoption of Battery Electric Vehicles (BEVs)), they found people from poorer areas drive shorter distances and tend to own petrol vehicles, and as a consequence create less overall road transport-related $NO_x$ and $PM_{2.5}$ compared with their wealthier counterparts. The average difference in vehicle age between the most and least wealthy households was just 1.2 years. This was thought to be due to the ages of vehicles in wealthier multi-car households and specifically the ages of their second and sometimes third cars. The UK National Travel Survey for 2016 (England) showed vehicles older than 10 years comprise 43% of cars owned by households with the lowest income quintile compared to 24% for the highest incomes. The proportion of vehicles 6 to 10 years old is evenly split between households by income.

Cairns et al. (2017) found distinct differences between car ownership and use in urban vs rural areas. Cars in urban households at the time of this study tended to be petrol rather than diesel powered, and urban drivers tended to drive less annual distance. These perspectives may be masked in national average figures. Spatial deprivation and social factors in vehicle ownership and use are not represented in network and link-by-link flow data that is usually used in air pollution modelling. More broadly the dynamic nature of the passenger transport fleet (for example uptake of hybrid and BEVs), and other mobility changes post-COVID can mean that conclusions drawn in even the recent past may not necessary be replicated in the present day.

## 7.3 Differentials in ambient air pollution impacted by occupation

Occupational exposure to air pollution in the workplace is covered under Health and Safety at Work legislation and fall outside of the Terms of Reference of AQEG. Nevertheless, there are many occupations and workers for which their workplace is the outdoor environment, and that their occupational exposure is determined to a degree by ambient air pollution concentrations.  These sections consider a small number of professions which are substantially impacted by outdoor pollution, however the coverage is by no means exhaustive.

### 7.3.1 Professional drivers and roadside workers

Professional drivers made up 3.6 per cent of the UK working population (2011 census), making it one of the largest occupational sectors in the country. This is likely to be an underestimate of the total number of people who drive for a living since it does not include drivers within the emergency services and other occupations requiring long-distance travel as integral to the role (Lim et al. 2021).

The London-based Driver Diesel Exposure Mitigation Study (DEMiSt) (Lim et al. 2021) explored the exposure of different driving occupations: taxi drivers, couriers, waste removal workers, heavy freight drivers, utility services, bus drivers, and emergency services providers. It made measurements of diesel exhaust exposure, using black carbon as a surrogate, on 141 London-based drivers based during their working days in 2018 and 2019. Their average concentrations were compared with those in outdoor settings. Individuals who drove as part of their job had an average black carbon exposure that was one third greater than the average concentration measured at the fixed monitoring site alongside the busy Marylebone Road and four times greater than the average concentration found in central London, as measured in North Kensington. Exposure times varied between different driver jobs. Taxi drivers had the longest on-shift driving time (6.5 hours) as well as the highest exposure levels while driving. Conversely, heavy freight drivers who had the second longest driving shifts (6.1 hours) had the fourth highest level of exposure. This difference most likely reflects periods spent travelling outside of the congested inner-city areas, compared to the exposures typically experienced by London’s taxi drivers.

Several drivers experienced high exposure events while driving in congested traffic, which remained high within the vehicle cabin for up to 60 minutes, even after they have had travelled away from pollution hotspots. Transits through Blackwall and Rotherhithe tunnels led to increases in black carbon concentrations that remained in the vehicle cabins for up to 20 minutes. Driver exposure was significantly lower on the weekends when there was less road traffic than during the weekday. The study also showed that exposure varied due to many factors: location, time of day, day of the week, wind and vehicle speeds, window position and background air pollution during the study, few of which can be controlled or adapted by the driver.

The London study is consistent with those in other countries e.g. a small study of seven New York taxi drivers (96 percent of taxi drivers in New York are migrant workers) found BC and $PM_{2.5}$ levels to be double that of background readings (Gany et al. 2017).

### 7.3.2 Outdoor workers

Studies of roadside workers also indicate that they are exposed to concentrations that are much higher than background air pollution concentrations. For example, in Almaty, Kazakhstan outdoor security guards had average wintertime $PM_{10}$ exposures of 360 µg $m^{-3}$. The authors considered that these exposures may be representative of many people doing outdoor jobs, which tend to be part of the informal job market, including outdoor market vendors (Vinnikov et al. 2020). Sehgal et al. (2015) studied exposures of toll booth workers in Delhi and compared these with booth staff at lesser trafficked locations including school and university entrances. The workers at busy central city locations had mean $PM_{2.5}$ of 219 µg $m^{-3}$, double that of workers in the least busy entrance booths.

### 7.3.3 Place of work or study

Liška et al. (2024) investigated exposures to $NO_2$, $PM_{2.5}$ and $O_3$ in the Central Belt of Scotland, stratified by age, sex, ethnicity and socioeconomic status (SES), the latter defined by the Carstairs index of deprivation. Pollutant concentrations were derived from the EMEP4UK model at 1.5 km spatial resolution and hourly temporal resolution. Using anonymised personal data and census data, they were also able to include the hours spent at place of work or study in the calculation of exposure; most studies are restricted to exposures at residential address only and/or do not have individual-level data.

*Socioeconomic Status*: The study showed a pattern of generally increasing exposure to $NO_2$ and $PM_{2.5}$ but decreasing exposure to $O_3$ as the level of deprivation increases. For example, median $NO_2$ was 3.2 µg $m^{-3}$ greater for those in the most deprived decile compared to those in the least deprived decile. This is consistent with the general tendency for areas of greater deprivation to be associated with higher density housing and roads where primary emission density is higher. However, the pattern across SES deciles was not completely uniform: a somewhat complex relationship between SES and residential exposure has also been noted elsewhere (e.g. Fecht et al. 2015, Temam et al. 2017), suggesting a relationship between air pollution and SES that can be area specific.

When exposure was calculated to include time spent at place of work or study, those in the least deprived decile have largest increase in median $NO_2$ exposure (8.5%) and largest decrease in $O_3$ exposure compared with exposure based on place of residence only. The patterns of exposure to $PM_{2.5}$ follow $NO_2$ but with smaller magnitudes. In other words, when time spent at place of work or study is accounted for, the differentials in exposure to air pollutants between least and most deprived are smaller than those derived using place of residence only. This statement is based on exposures on average in an SES decile; clearly some individuals may not work or study elsewhere or may work or study in locations of higher or lower pollution than that experienced on average for individuals in their SES decile.

*Age*: Median exposure to $NO_2$ was greatest in the 21-30 age range, sharply increasing from the lowest median exposures in childhood before slowly decreasing again through latter years, similar to that reported by Mitchell and Dorling (2003) and Barnes et al. (2019). However, the largest increase in median exposure to $NO_2$ when including place of work or study was for the 31 - 55 age group. Both these patterns are consistent with the tendency of young adults to live and work in urban centres, where $NO_2$ concentrations are already high, before tending to move outwards in later life but still to commute into urban areas for work. However, even with incorporation of place of work in the exposure calculation, the 31 - 55 age group exposure to $NO_2$ was still lower, on average, than that of the 21 - 31 age group. The patterns for exposure to $O_3$ with age were reversed compared with those for $NO_2$, whilst the pattern in exposure to $PM_{2.5}$ was similar to $NO_2$ but with smaller differences in exposures across the age groups and smaller changes when accounting for place of work.

*Sex*: There were no differences in median exposures with sex for any of the pollutants, but for all three pollutants, male exposures were more affected (on average) by time spent at workplace than were female exposures (median exposure to $NO_2$ increased, median exposure decreased for $O_3$, very small median increase for $PM_{2.5}$).

*Ethnicity*: Based on place of residence only, the White ethnic group had substantially lower $NO_2$ exposure (difference in median exposure of 3.6 µg $m^{-3}$), and slightly lower $PM_{2.5}$ exposure, than the minority ethnic groups, but highest exposure to $O_3$. This is due to minority ethnic groups predominantly living in the cities and towns and is consistent with Fecht et al. (2015) who found that in England and the Netherlands, at regional level, neighbourhoods with <20% non-White ethnic individuals had lower concentrations of $NO_2$ than those with>20% non-White ethnic individuals. However, when time spent at place of work or study was included the White group experienced the greatest increase in $NO_2$ exposure and greatest decrease in $O_3$ exposure compared with exposure based on residence only.

In summary, the inclusion of workplace in estimates of exposure attenuated some of the exposure differentials associated with socioeconomic status, ethnicity and age observed in exposure assessments based only on place of residence.

## 7.4 Evidence on transport interventions

There is substantial evidence that air pollution exposure varies by different modes of transport. However as identified in AQEG (2022) the vast majority of in-cabin studies took place in a period when exhaust abatement technologies were less mature and before the advent of widespread battery electric vehicles. There is a need for new studies to consider current and future differentials in air pollution concentrations and exposure by transport mode. These should also include exposure to non-exhaust emissions.

Studies that combine transport use with air pollution exposure by mode, and the socioeconomic context of those travelling, are necessary to evaluate differentials in air pollution exposure due to transport, and by extension help design and evaluate possible interventions. These will require the integration of exposure by mode and the social economic metrics of travellers into transport models.

Mitsakou et al. (2021) suggested a hierarchy of transport-related air quality interventions to reduce exposures: the prevention of emissions, mitigations such as mechanical ventilation for public transport and careful positioning of bus stops, and the shift to active travel, especially active travel that is physically separated from roads.

Several intervention types have focused on air quality and transport at schools and the areas around them. These include ‘school streets’ that restrict motorised traffic near schools at pick-up and drop off times and open these areas for play and active travel. Gellatly and Marner (2021) measured air pollution before and after the implementation of 16 school streets in London. They reported that $NO_2$, one of the pollutants from traffic, was reduced by 23% and the number of children walking or cycling to school increased by 18%. A review of schemes by Davies (2021) concluded that, with careful design, school streets could lead to traffic reductions over a wider area.

Further opportunities for interventions are likely to arise out of analysis from integrated exposure, socioeconomic and transport models. For example, Gutzman et al. (2013) suggest that opportunities do exist, but they are unlikely to be identified without considering specific situation and setting.

The British Safety Council have called for employers to take more steps to protect outdoor workers from air pollution and for more action to reduce sources and exposure. Within their call they provided examples of affected occupations including street cleaners, refuse workers, traffic police, cycle couriers, construction workers, maintenance workers, newspaper sellers, gardeners, teachers and security guards (British Safety Council, 2024).

# Chapter 8 - Policy influence on air quality differentials

## 8.1 Introduction

An objective of the 25 Year Environment Plan (Defra, 2018) is "*to ensure an equal distribution of environmental benefits, resources and opportunities*". The Plan also describes seeking "*to improve social justice by tackling the pollution suffered by those living in less favourable areas*". Within the UK inequality is often inherent in the urban fabric and this changes only slowly over decades. For example, historically workers houses were in the east of cities while affluent houses were in the west to avoid pollution due to the prevailing wind direction. This urban fabric with larger lower density houses in the west and smaller higher density houses in the east is still present in many cities.

Differentials in exposure to poor air quality, or to the burdens of regulation, are also the composite result of more contemporary policies and processes. National policies on land use, economy, energy, and transport combine with current and historic decisions by local government and individuals to shape how and where people live and spend time (Barnes et al. 2019; Brainard et al. 2002). Unpicking cause and effect is impractical and outside the remit of AQEG.

Many policy-driven changes might have inequitable effects on air pollution exposure in the future. In particular, delivery of net zero policies will require substantive changes to personal transport and how individuals heat their homes. Economic and social factors make it unlikely that uptake of lower-emission technologies will be evenly split across society, with resulting consequences for the distribution of air pollution emissions; although some research does suggest that pursuit of 2040 carbon emission targets through electrification of the fleet will lead to greater reductions in absolute differentials in exposures experienced by more deprived areas in England (Woodward et al., 2024).

Inequity might relate to either:

- how positive and negative features are distributed among different members of society. This is termed 'distributive justice' (Stephens and Church, 2017). Most of this AQEG report has been concerned with the distributive justice of exposure to poor air quality;
- how different groups are affected by policy choices, such as burdens imposed by actions to improve air quality. This has been termed 'policy justice' (Stephens and Church, 2017); or
- how people are able to engage with, and affect, decision-making. This has been termed 'procedural justice' (Stephens and Church, 2017). In principle, distributive and policy justice are not dependent on procedural justice; individuals need not always be part of a decision to benefit from it. In practice, however, procedural justice is widely seen as both a requirement in and of itself, and a means to achieve policy and distributive justice.

Stephens and Church (2017) also define the concepts of 'intranational justice' and 'international justice', which view distributive justice over different spatial scales, and ‘intergenerational justice’, which is increasingly relevant in the context of long-term planetary harm and public health threats.

Justice implies ‘fairness’, but fairness is subjective. Mitchell (2019) notes that distributive inequality need not always imply unfairness, and that people disagree on whether many causes of inequality are fair or not. This also links to the debate on the relative fairness of equality of action vs equality of effect. Interpretation of ‘equality’ to mean equal *treatment* (as opposed to delivering equal *outcomes,* also known as ‘equity’) has shaped how some UK policy has been implemented (Bristow, 2021), with implications for air quality exposure distribution.

Taken at face value, most air quality policy tends to apply equally to all members of society irrespective of their social group, but also irrespective of their specific needs. Whether this is deemed fair may depend on an individual’s concept of justice. Regulatory philosophy is thus a potentially divisive political issue. Different groups may have contrasting beliefs about the role of distributional considerations in decision making, about the relative importance of progressive vs regressive policy, and ultimately about what is ’fair’. Irrespective of concepts of “fairness”, not accounting for differentials in exposure to air pollution related to socioeconomic status in managing air quality will likely lead to policy inefficiencies as it fails to recognise that in more deprived communities, per person per unit of exposure will lead to greater damage and associated health costs than for those less deprived (Woodward et al., 2024).

Regardless of how individuals view fairness, and irrespective of any additional societal costs caused by the distribution of poor air quality, a perception of inequity may create additional barriers to public engagement with actions to improve air quality. O’Beirne et al. (2020) concluded that inadequacy of current procedural and policy justice mechanisms for greenhouse gas reduction (e.g. Povitkina et al., 2021, CCC, 2022) manifests in members of the public as disillusionment, scepticism, and lack of interest. The required actions to improve air quality, and the direct implications of those actions, are often moving from large corporations to individuals (AQEG, 2024b). Delivering these actions is likely to be easier if they are perceived as fair. The financial costs to individuals of local air quality actions, such as the London ULEZ, are an important part of the political debate around the appropriateness of these actions. Objections often centre on groups who feel they are disproportionately negatively affected. An absence of clear narrative of how different groups have been considered in decision-making may create a void for misinformation to enter discussions (AQEG, 2024b).

Access to air pollution data and information may not necessarily be equitable. For example, research indicates that those accessing digital sources on air pollution may be younger (<36 years), male and non-white (Schulte and Hudson, 2023). Whilst it is acknowledged that access to information does not necessarily lead to behaviour change, ensuring communications campaigns are actively inclusive rather than passive can increase awareness and facilitate policy implementation. At the time of writing, Defra are undertaking an in-depth review of the Daily Air Quality Index and Air Quality Information Systems more generally, hence this is not covered further in this report.

## 8.2 Effects of national air quality policy

National-level policy specific to improving air quality is expansive and not set out in detail here. Very broadly, it seeks either to:

a) reduce emissions, on aggregate or from specific sources;
b) ensure that concentrations remain below prescribed values; and/or
c) reduce population-weighted concentrations.

Air quality objectives (Air Quality (England) Regulations (2000) (as amended)) are set as concentrations which take account of potential effects on sensitive subgroups of the population, but they apply equally to all outdoor locations where members of the public have regular access. They do not apply to places of work, which are controlled by the Control of Substances Hazardous to Health (COSHH) Regulations 2002 (as amended). Workplace exposure limits are set to be practical even in heavy industrial activities, meaning that vulnerable individuals in non-industrial work settings have relatively limited legal protection from poor air quality (Marner and Laxen, 2016) (see chapter 7).

Air quality limit values (Air Quality Standards Regulations (2010) (as amended)) apply to defined outdoor locations. The annual mean concentration target set in the Environmental Targets (Fine Particulate Matter) (England) Regulations 2023 only legally applies at monitoring stations run by Central Government, but Defra considers it to apply wherever people may be regularly exposed for long periods. The population exposure reduction target (PERT) in the 2023 Regulations has been designed to reduce the average $PM_{2.5}$ concentrations experienced by most people. Beyond these differences, there is no intrinsic requirement for social characterisation when designing measures to improve air quality. That said, Woodward et al. (2024) have estimated that achieving the 2040 PERT for $PM_{2.5}$ will likely lead to further reductions in the differential in exposure experienced by more deprived areas when evaluated using the IMD as a measure of deprivation, beyond those achieved through net zero policies.

Using two policy scenarios: emission reductions across all sectors in addition to measures towards reaching NZ (PERT2040) and a greater focus on reducing urban sources of primary $PM_{2.5}$ (PERTUrban2040), resulted in a modelled reduction in the absolute differential in exposure experienced by more deprived areas in England relative to 2018 of 43% and 59% respectively (cf. 37% reduction in population exposure), suggesting benefits can be maximised by targeting urban sources of primary $PM_{2.5}$. More broadly decarbonisation actions aligned with net zero targets for 2050 offer many opportunities for air quality improvement and the reduction in dipartites in exposure to pollution (Royal Society, 2021).

## 8.3 Effects of local air quality policy

Implementation of air quality policy is frequently a matter for local government, historically focusing on achieving objectives, or in some cases limit values. The 2023 Air Quality Strategy sets out responsibilities for local authorities which go beyond targeting concentration hot-spot locations. This partly reflects the implementation of the PERT for $PM_{2.5}$ but also highlights specific consideration of sensitive users irrespective of any exceedance of an objective, limit value, or target.

The August 2022 update to the Local Air Quality Management (LAQM) Policy Guidance (Defra, 2022) builds on 'lived experience' workshops undertaken by the Environment Agency with communities and representative groups and policy makers to include new responsibilities for local authorities to take account of air quality disparities. Local authorities should consider disparities resulting from differential exposures to air pollution, as well as the implications of policy implementation for those with least ability to limit their exposure or change their behaviours (e.g. through limited choices over where they live, work, travel or heat their homes) (Figure 8.1). Various avenues to address air quality disparities are outlined in the guidance (Figure 8.2), including working across policy disciplines and implementing inclusive communication strategies to ensure that they engage communities that are not well-represented in air quality decision making processes effectively.

| | |
|---|---|
| **Employment** | Many workers can't choose to work from home or travel at non-peak times to avoid congestion. Certain roles are likely to be more exposed to pollution sources and may have limited choice over the work they do. |
| **Housing** | Those living on lower incomes may be limited in what changes they can make to their homes, also in their choice of where to live. Moving home to reduce exposure is unlikely to be a simple option. |
| **Fuel-poverty** | Those living in fuel-poverty are limited in their choice of how, when and with what they heat their homes. These people are also likely to be less willing to well ventilate their homes due to the loss of heat. |
| **Travel** | Those living on lower incomes are likely to have less choice over their transport options and reduced capacity to change how they travel. Those who drive may be less able to upgrade their vehicle. |
| **Schools** | Those living on lower incomes are likely to have less choice over where they live and where their children go to school and consequently the air pollution they are exposed to both whilst travelling to school and whilst there. |
| **Access to green space** | Individuals from deprived communities generally have less access to quality green space close to where they live. Green spaces can provide locations for people to congregate, relax and exercise further from sources of traffic pollution. Access to green space helps to achieve multiple health objectives. |

*Figure 8.1. Limitations of choice (adapted from LAQM (Defra, 2022) page 22)*

| | |
|---|---|
| **Community participation** | Community participation aims to facilitate the involvement of local residents and is based on the belief that those who are affected have the right to be involved in the decision-making process. |
| **Planning** | Protection vulnerable populations such as those in day care centres, schools and hospitals from poor air quality and exposure impacts. Mitigate environmental risks in disadvantaged areas and to ensure balanced distribution of air quality burden across a local authority, avoiding the accumulation of environmental deprivation and pollution hotspots in specific areas. |
| **Transport planning** | The benefits and negative impacts of transport are not evenly spread across society. A sustainable transport system will not only reduce vehicle emissions, but will also help to mitigate health and mobility disparities and improve social interactions, liveability and amenity values. |
| **Housing departments** | Individuals in rented accommodation may be limited in the choices they can make when it comes to improving or modifying their properties to decrease the concentrations of indoor air pollution. Local authority housing teams have an important role to play, not just in offering guidance and support to both tenants and landlords, but in encouraging higher standards in rental accommodation. |
| **Open / Green space** | Public and accessible open and natural spaces provide air quality, social and health benefits through nature and ecosystem functions. They also have recreational and cultural functions and can provide social gathering and meeting spaces. |
| **Pollution control and environmental protection** | Actions may include, zoning approaches and functional restrictions, promotion of active and public transport choices, clean energy programmes and careful siting and control of hazardous activities and polluting industries. |

*Figure 8.2. Local Authority avenues to address Air Quality Disparities (adapted from LAQM (Defra, 2022) page 22)*

## 8.4 Transport policy

Road transport emissions have historically been a significant source of poor air quality in urban areas and close to other major roads.  There is a significant amount of policy which influences road transport, from national strategies to local implementation.  Furthermore, policy aimed at decarbonising the transport network (e.g. DfT, 2018) will also have implications for exhaust emissions of air pollutants.

Many of the opportunities to reduce road transport emissions vary geographically.  Cities which benefit from significant historic investment in public transport infrastructure can rely more heavily on these modes if there is existing capacity.  More remote communities often have fewer opportunities other than private cars.  There are also significant regional differences in the existing use of active travel; for example, residents of the West Midlands travel almost 50% less distance by walking or cycling than residents of London (Figure 8.3). The reasons for this are complex and not explored here, but it demonstrates the need for spatially targeted transport policy.  An important feature of transport planning is thus the requirement for local transport authorities to produce Local Transport Plans, following guidance produced centrally (e.g. DfT, 2009; DfT, 2022).

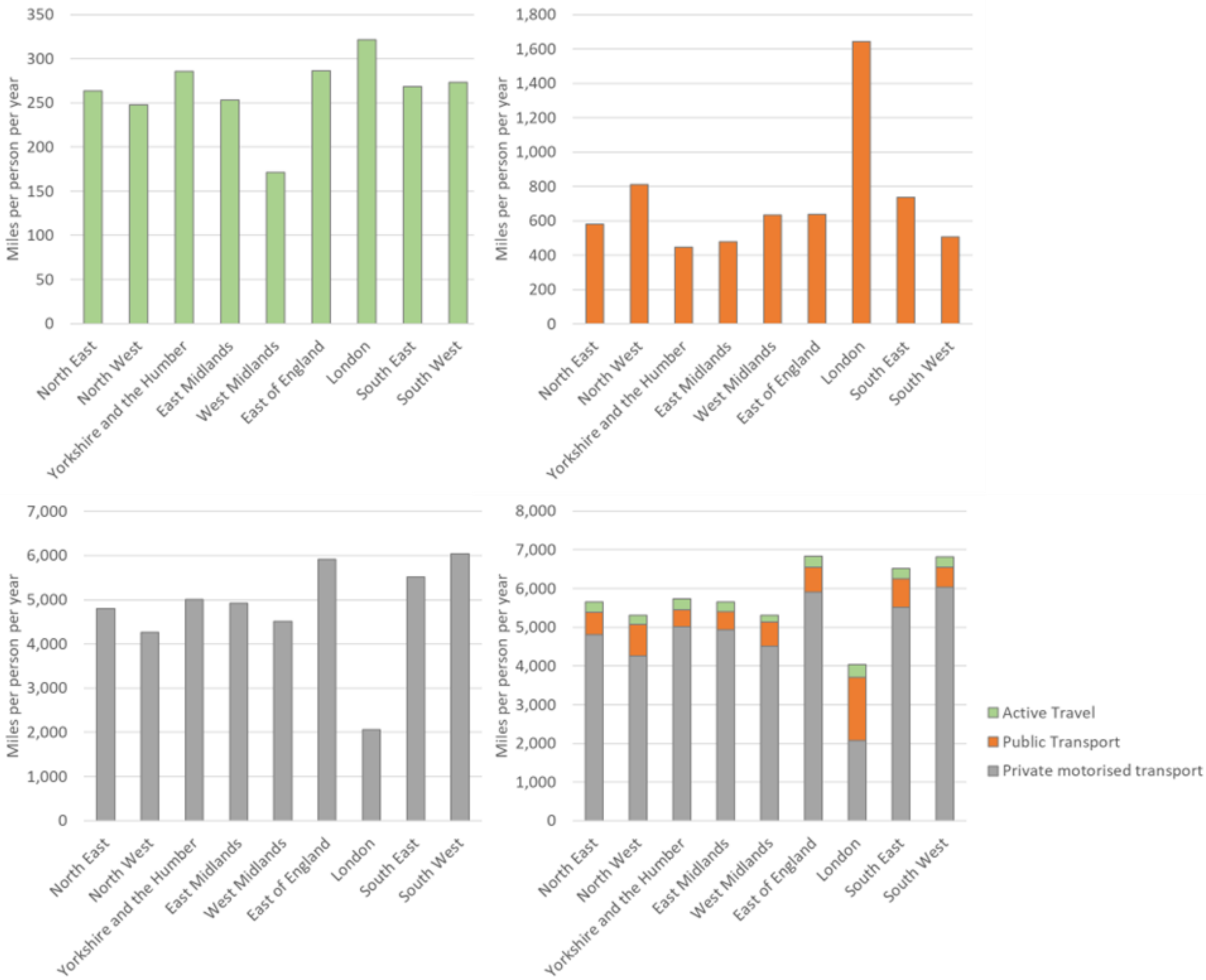


*Figure 8.3. Average distance travelled by mode and region of residence in year ending June 2023 (data from Department for Transport, 2024).*

A key set of transport policies (leading to regulations) which are not location-specific, and which have driven improvements in urban $NO_2$ concentrations are the European type-approval vehicle emissions standards. (AQEG note however that improvements over time have not been smooth, with some time periods seeing faster improvements than others, with particular problematic issues around the underperformance of Euro 5 diesel passenger cars.) LAQM and local transport policy have often relied heavily on the efficacy of these standards. Where there have been failures in LAQM (e.g. Moorcroft et al., 2013, Barnes et al., 2014) these have disproportionately failed more deprived areas (Mitchell et al., 2015, Barnes et al., 2019). Mitchell et al. (2015) studied the benefits of environmental policy directly on $NO_2$ and $PM_{10}$, by investigating how the relationship between exposure and deprivation changed between 2001 and 2011. They reported that UK small area analyses showed that in 2001 poor air quality was much more prevalent in socioeconomically deprived areas, but extended the analysis to consider how this changed between 2001–2011, a period when significant efforts to meet EC Air Quality Directive limits were made, and some air quality metrics improved. The study found that air quality improvement was greatest in the least deprived areas, whilst the most deprived areas bore a disproportionate and rising share of poor air quality including non-compliance with air quality standards. The period studied also corresponded with an increase in $NO_2$ concentrations in the mid-2000s at roadside locations due to an increase in diesel vehicles resulting from Vehicle Excise Duty reductions to meet carbon reduction targets (Barnes et al., 2018). As Mitchell et al. (2015) indicates, areas with the highest levels of deprivation were likely to have been worst affected by this inconsistency between air quality and energy efficiency policies.

Namdeo and Stringer (2008) examined road user charging (RUC) scenarios in Leeds using a model to predict $NO_2$ and comparing each scenario with a 2005 'base' case. The study correlated $NO_2$ concentrations with derived indices of social deprivation and health, concluding that a positive but weak relationship existed between air quality and social deprivation, and that deprived population groups were disproportionately exposed to higher $NO_2$ concentrations. The relationship between air quality and health status of the population was also weak, although there was a strong relationship between social deprivation and health status. The study concluded that in contrast to the study by Mitchell et al. (2015) described above, the RUC scenarios reduced the $NO_2$ exposure disparity between affluent and deprived populations.

Williamson et al. (2021) concluded that where actions to tackle transport emissions in London were ultimately successful, those exposed to the highest concentrations experienced the largest benefits. This was supported by Woodward et al. (2024) who estimated that reductions in the absolute bias in exposure towards less deprived areas in England were also seen in London at 32% and 59% relative to 2018 for PERT2040 and PERTUrban2040, respectively. However, Brook et al. (2023) regarded that existing air quality and transport policy was unlikely to deliver any significant change in the relative distribution of concentrations across deprivation levels in London up to 2030, thus not resolving existing inequalities (see Chapter 5).

## 8.5 Development of land management policy

The 1990 Town and Country Planning Act and 2004 Planning and Compulsory Purchase Act govern the planning process for most development in England; planning decisions are made by local authorities, although decisions may be called in by, or appealed to, the relevant Secretary of State (SoS). The 2008 Planning Act sought to streamline the process for Nationally Significant Infrastructure Projects (NSIPs) which, following the 2011 Localism Act, are now decided directly by the SoS. SoS decisions are informed by recommendations of planning inspectors appointed by the Planning Inspectorate, although the SoS is not compelled to agree with those recommendations.

The National Planning Policy Framework (DLUHC, 2023) sets out the Government's planning policies for England and how these should be applied. It accompanies a series of National Policy Statements for specific types of infrastructure. These overarching documents are supported with linked guidance (e.g. the National Planning Policy Guidance) and also cascade down to local planning policy. Parallel to this, the 2017 Town and Country Planning (Environmental Impact Assessment) (EIA) Regulations set out specific requirements for the assessment of certain schemes.

The Levelling-up and Regeneration Act 2023 includes powers to change how development decisions are made, including amending the 1990 Town and Country Planning Act with respect to certain planning functions. It is too early to determine how this will affect decision-making, air quality, or any resulting impacts on specific groups.

### 8.5.1 Air quality considerations in development policy

Significant volumes of air quality assessment guidance is published by various bodies to cover different types of interventions in different regimes. Some of this requires explicit consideration of differential effects, as shown in Table 8.1, however the focus of most air quality assessments to inform development decisions remains the universal achievement of the air quality objectives and not whether the distribution of air quality impacts is equitable.

Requirements for measures to improve air quality within, and caused by, new developments have predominantly been linked with exposure to $NO_2$. As exceedances of the annual mean $NO_2$ objective become less frequent, the perceived need for such measures is diminished. At the same time, greater public recognition of poor air quality may increase the commodification of personal exposure. There is thus the potential for increased air quality inequity linked to affordability, driven by both proximity to emissions sources, ambient concentrations and development design (e.g. Ferguson et al., 2021). Future guidance on applying the Fine Particulate Matter Regulations (2023) is expected to more universally promote better design but is unlikely to address this directly.

*Table 8.1. Examples of air quality assessment guidance which requires consideration of inequalities*.

| Assessment | When used | Description |
|---|---|---|
| Transport Analysis Guidance (TAG) unit A4.2 (distributional impact appraisal) | New transport schemes | Considers the expected air quality (and other) impacts experienced by households with different levels of income as well as considering attractors which might affect different social groups. Intended to address the requirement of the Green Book (see Section 8.7, below). |
| Design Manual for Roads and Bridges (DMRB) LA 112 (population and human health) | National Highways schemes | Considers health profiles for affected communities, including prevalence of pre-existing health issues, long-term illness or disabilities, life expectancy and income deprivation. |
| Institute of Air Quality Management Planning for Air Quality | Planning applications | Encourages detailed consideration of locations where particularly sensitive members of the population are likely to be present in areas where pollution concentrations are high. |
| Institute of Environmental Management and Assessment (IEMA) guidance for determining significance for human health in EIA | EIA | Considers the sensitivity of receptor groups within a study area, taking account of socioeconomic health determinants and considers the impacts of a scheme on aspects including air quality. |

The Green Book (HM Treasury, 2022) (see Section 8.7 below) provides a framework which can be used to appraise the socioeconomic effects of new developments but is typically only used where Central Government funding is sought.

### 8.5.2 Planning and development decision-making

An underlying aim of development management policy is often to conserve existing features and characteristics. This may entrench existing differentials and disparities since developments are favoured where they align with current land uses. Elements of planning policy have also been interpreted as seeking to conserve ways of life which are not traditional to minority groups (e.g. Beebeejaun, 2004).

New development often generates ‘winners’ and ‘losers’, both in perception and in the reality of environmental and economic consequences. Irrespective of a perceived national need for a development, individuals seldom wish to be disproportionately negatively affected themselves (e.g. Vittes et al., 1993).

Hunold and Young (1998) considered how decisions were made regarding the siting of hazardous industrial facilities in the US and Europe and concluded that community engagement in decision-making remains preferable to a more authoritarian imposition of interventions, even where the latter objectively minimises total exposure. Significant weight is given to public engagement in English planning, from initial consultation through to public inquiries and hearings.

Neighbourhood planning also gives local communities direct planning powers. However, while the overall concepts of public engagement and community-led decision making seem fundamentally egalitarian, effective engagement can require significant time, knowledge and resources. These processes may thus amplify voices that already hold power within society (Bristow, 2021, Parker et al., 2023). Carrick et al. (2023) describe a “*paradox of participation*” whereby ineffective engagement can be counter-productive in all relevant respects.

Planning can also be strongly adversarial and the manner of engagement of local groups, and indeed local planning authorities, can be influenced by the prospect of considerable costs being awarded against a ‘losing’ side. These issues are not specific to air quality but nevertheless can serve to differentially empower and disempower certain groups.

Bristow (2021) reviewed elements of the English planning system with respect to racial inequalities and found a clear focus on equality of treatment as opposed to equality of outcomes, with policy thus continuing to reinforce existing disparities and discrimination within the planning system.

The Town and Country Planning Association carried out a review of planning in England (TCPA, 2018). One highlighted aspect was disbenefits associated with the relaxation of permitted development rights. There are implications for air quality exposure of using permitted development rights to convert buildings into residential dwellings without considering their suitability with respect to air quality. Bristow (2021) suggests that this may be of particular significance for “BAME communities” who are more likely to live in poor quality accommodation. Similarly, planning issues related to the provision of affordable housing have clear implications for where, and how, different groups are exposed to air quality.

## 8.6 Other relevant existing policy

Other pre-existing policy which might affect differentials and disparity in air pollution exposure is wide-ranging and not restricted only to policy with explicit environmental protection goals. A complete list is beyond the scope of this chapter, but some relevant instruments are described below.

The Aarhus Convention (UNECE, 1998) provides three pillars:

- Access to information (Articles 4 and 5);
- Public participation (Articles 6 to 8); and
- Access to justice (Article 9).

The Aarhus Convention underpins many principals by which the public can engage with policy making. The UK is party to the Convention treaty but its full status in current UK law is complex.

The Equality Act 2010 protects people from discrimination in the workplace and society. Part 1 of the Act: ‘Socioeconomic inequalities’ was never brought into force in England, where the Act focuses on nine ‘protected characteristics’ including age and race but not economic factors. Positive action is encouraged by the Equality Act, but positive discrimination is prohibited. Interventions which selectively affect protected groups may however conflict with the Act.

The Levelling-up White Paper (2022) set out a plan to work towards resolving inequity caused by geographical disparity within the UK. The white paper set 12 levelling-up “*missions*” covering living standards, public investment in Research and Development, public transport and digital connectivity, education, skills and training, health, well-being, pride in place, housing, crime, and local leadership. The Levelling-up and Regeneration Act (LuRA) became law in 2023. Part 1 of the LuRA centres around the levelling-up missions, but the main elements provide changes to the planning system (see above). The former Department for Levelling Up, Housing and Communities (DLUHC) explained that the LuRA would encourage developers to build, making it easier to gain planning permission and cutting “*EU-red tape*” on environmental assessment (DLUHC and Gove, 2023).

## 8.7 Air quality and new policy design

The Green Book (HM Treasury, 2022) provides guidance on appraising new policies, programmes and projects, which aims to optimise the social value of public spending. It can be used for new legislation or non-legislative policy change, as well as spending proposals (for example publicly funded infrastructure). The Green Book may be used to appraise policies across all levels of government but is not always used in full to appraise local policy.

In 2020, Mayors from cities across England called for reforms to the Green Book, which it was claimed reinforces regional inequality by skewing investment toward already prosperous parts of the country (Financial Times, 2020). However, Breach and Jeffrey (2020) concluded that the issues were more complex than this, and unlikely to be solved through revised

Green Book financial metrics. They argued strongly that the Green Book should continue to be used to appraise local, as well as national, policy.

A review of the 2020 Green Book (HM Treasury, 2020) considered the levelling-up agenda in some detail and advised that appraisers consider with care how different groups will be affected by interventions. Annex A3 of the current Green Book covers distributional appraisal, defined as "*the assessment of the impact of interventions on different groups in society*". While the focus is on household income, appraisals should take account of unintended collateral effects that may unfairly impact particular parts of the UK, or groups within UK society. The accompanying Magenta Book (HM Treasury, 2020b) explains that evaluation should include social costs such as employment, health, wellbeing and productivity.

As is shown in Chapter 5 of this current AQEG report, these distributional effects go beyond classification by income alone. Many, if not all, policies have the potential for multi-dimensional effects; for example, impacting individual's incomes, health, exposure settings, and emissions activity. Providing a complete appraisal of all potential pathways before implementing a new policy is often likely to be impractical. The Green Book explains that the level of detail and complexity should be proportionate to the likely impact on those affected.

Two recent examples of new air quality legislation for England are The Air Quality (Domestic Solid Fuels Standards) Regulations 2020 and The Environmental Targets (Fine Particulate Matter) Regulations 2023. In both cases, an Impact Assessment (IA) was carried out to follow Green Book guidance. The IA for the 2020 Regulations considered distributional impacts, including financial impacts on fuel poor households and impacts on both fuel suppliers and air quality in rural areas; the air quality assessment was restricted to a stated expectation of a strong positive impact in rural areas (Defra, 2019). The IA for the 2023 Regulations considered the impact on deprivation by overlaying predicted maps of $PM_{2.5}$ concentrations onto IMD data and showing that there would be an overall reduction in the range of population weighted mean concentrations when comparing different deciles (Defra, 2022). While the distribution of societal costs of actions required to meet the 2023 Regulations was not considered, it is expected that any significant policy measures pursued to reach the target will be subject to their own IA. (See also Woodward et al. (2024) for subsequent analysis of the impacts of the 2023 Regulations on disparities using IMD.)

The Integrated Impact Assessment of the London-wide ULEZ (Jacobs, 2022) also considered distributional impacts. This considered how changes in air quality would be distributed across different IMD deciles, concluding that benefits would be evenly distributed across different communities. It also considered car and van ownership rates by income deciles and highlighted potential adverse financial impacts on socioeconomically deprived car owners. Similarly, the Joint Air Quality Unit (JAQU) advises that local authorities implementing Clean Air Zones should undertake distributional analysis across a range of socioeconomic groups (including income, age, gender, ethnicity and disability) as part of their Business Case to inform boundary setting and differential charging. Even so, many people have argued that CAZs and the ULEZ expansion have disproportionately impacted deprived communities, although evidence does not necessarily support this with those most adversely affected by air pollution being net beneficiaries of reductions in traffic emissions (Chamberlain et al., 2023; Greater London Authority, 2023; Liu et al., 2023; Rashid et al., 2021).

Robinson et al. (2016) considered how distributional analysis affected US regulation prior to 2016. They found that despite a requirement, stemming in the US from Presidential Executive Orders, for decision-makers to consider distributional impacts, sufficiently detailed analyses to inform these decisions were rare. The difficulty in carrying out, and appropriately acting on, distributional appraisal meant that analysis was often replaced with cursory and unevidenced statements of compliance.

The IAs of the 2020 and 2023 Regulations described above, and the assessment of the London-wide ULEZ, suggest that the UK situation is not the same as that reported by Robinson et al. (2016) for the US, but it is also not clear in these examples whether the current Green Book guidance was followed to its full extent. This is likely to reflect the potentially extreme complexity that a complete appraisal of multi-pathway distributional effects could entail. It is also unclear the extent to which the assessments which were carried out informed the policies' designs.

Parallel to the requirements of the Green Book, historically there was also a legal requirement for public sector organisations to carry out Equality Impact Assessments (EqIAs) of their policies and functions. These are now voluntary, but EqIAs carried out under the Equality Act 2010 nevertheless often form part of strategic decision making, such as for Local Plans. Some local authorities are extending these to include socioeconomic status, despite this part of the Equality Act not being enforced (e.g. Greater Norwich Councils, 2020). It is common, however, for EqIAs to lack depth and not be afforded significant weight in overall decision making. The Town and Country Planning Association (TCPA, 2019) has produced guidance attempting to move EqIAs "*beyond box-ticking*".

## 8.8 Future approaches to improving air quality

Several UK studies demonstrate that those responsible for the generation of traffic-related air pollution are usually the least marginalised communities, e.g. more affluent households (Barnes et al., 2019) or (in a study based in Barcelona) middle-aged European men (Cubells et al., 2024). Policies can be conceived of that acknowledge this unequal distribution of responsibilities and direct emissions reduction strategies at those households/behaviours that are generating the pollution.

Bristow (2021) set out several recommendations for improving air quality outcomes through the planning system, including an obligation for local authorities to carry out, and act on the findings of, EqIAs in their policy making. Carrick, et al., (2023) outlined how they believed public engagement might be improved, with key themes being transparency and the demonstrable empowerment of participants. While not specific to environmental inequality, TCPA (2018) provides a series of 24 recommendations for the planning system in England, including harmonising strategic planning approaches across England, enhancing the role of local planning, and greater accountability for local planning authorities.

Mitchell (2019) suggested various options to tackle environmental inequality (Table 8.2). A common approach is via community regeneration projects, but these tend to have many objectives and follow up evaluation with respect to environmental equity has been weak.

*Table 8.2. Possible Responses to Environmental Inequality (adapted from Mitchell, 2019)*

| Possible response to environmental inequality | Possible problems |
|---|---|
| Direct environmental hazard away from minority communities | Fewer local economic opportunities/jobs; more significant environmental damage elsewhere |
| Good neighbour (hazard v community) agreements | Risks 'greenwash', few sanctions, hard to police and enforce |
| Provide compensatory benefits to accept environmental hazard | Evaluation, cost, little culture of compensation |
| Invest in environmental regeneration in minority communities | Environmental gentrification |
| Raise environmental performance generally | Passive and effect on inequality unproven |
| Embed social justice appraisal in tools such as Strategic Environmental Assessment and Health Impact Assessments to avoid exacerbating inequality | Assessments exists but are advisory and can have limited effect on decisions |
| Litigation | Access to justice presents a barrier, especially uncertain costs |

Robinson et al. (2016) concluded that thorough assessment of distributional impacts is a prerequisite of good regulatory decisions. They highlighted that only by making clear the key trade-offs associated with a policy, can informed decisions be made. As set out in Section 8.7, the Green Book provides guidance on how this information may be provided, but the complexity of the issues is such that thorough appraisal is seldom possible, and it is often unclear how this information feeds into policy design. Better understanding of the complex ways in which interventions might affect different groups, such as provided in this current report, might empower more informed policy making. It seems likely that additional effort spent understanding any inequitable effects of policies, and making use of this information in policy development, would be helpful in meeting the social justice goals defined in the Defra 25 Year Environment Plan.

Air quality actions increasingly require public 'buy-in'. It seems likely that additional effort spent addressing any inequitable effects, and communicating how this has been done, might support delivery of these actions, albeit that individuals who are differentially affected may disagree on what is important and 'fair'. Including the public in the decision-making process, taking onboard their lived experiences to inform policy development, could help to mitigate feelings of disenfranchisement and lead to more effective implementation and equitable outcomes.

Decisions made at all levels have the potential to either increase, or reduce, inequity. Similar principals apply when setting national policy as when making decisions on individual developments. The issues are complex and multi-dimensional, with significant scope for unintended consequences. It seems that there is often a relatively poor understanding of how policies, actions, and projects will affect different groups, which limits the extent to which informed decisions can be made.

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