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Impacts of net zero policies on air quality in a metropolitan area of the United Kingdom: Towards world health organization air quality guidelines

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ABSTRACT

Climate change and air pollution are closely interlinked since carbon dioxide and air pollutants are co-emitted from fossil fuel combustion. Net Zero (NZ) policies aiming to reduce carbon emissions will likely bring co-benefits in air quality and associated health. However, it is unknown whether regional NZ policies alone will be sufficient to reduce air pollutant levels to meet the latest 2021 World Health Organisation (WHO) guidelines. Here, we carried out high resolution air quality modelling for in the West Midlands region, a typical metropolitan area in the UK, to quantify the effects of different NZ policies on air quality. Results show that NZ policies will significantly improve air quality in the West Midlands, with up to 6 μg m^{-3} (21%) reduction in annual mean NO_2 (mostly through the electrification of vehicle fleet, EV) and up to 1.4 μg m^{-3} (12%) reduction in annual mean $PM_{2.5}$ projected for 2030 relative to levels under a "business as usual" (BAU) scenario. Under BAU, 2030 $PM_{2.5}$ concentrations in most wards would be below 10 μg m^{-3} whilst under the Net Zero scenario, those in all wards would be below 10 μg m^{-3}. This means that the ward averages in the West Midlands would meet the UK $PM_{2.5}$ of 10 μg m^{-3}target a decade early under the Net Zero scenario. However, no ward-level-averaged annual mean $PM_{2.}$concentrations meet the 2021 WHO Air Quality guideline level of 5 μg m^{-3} under any scenario. Similarly for NO_2 only 18 wards (8% of the region's population) are predicted to have NO_2 concentrations below the 2021 WHO guideline level (10 μg m^{-3}). Decarbonisation policies linked to Net Zero deliver substantial regional air quality benefits, but are not in isolation sufficient to deliver clean air with air pollutant levels low enough to meet the 2021 WHO guidelines.

1. Introduction

Climate change is a long-term shift in global mean temperatures and weather patterns (Aguado and Burt, 2013). Climate change causes adverse effects such as flooding, heatwaves, biodiversity loss and rising sea levels (Abbass et al., 2022). Climate change is primarily caused by human activities (IPCC et al., 2022), e.g. combustion of fossil fuels (e.g. coal, oil and gas) with increasing emissions of long-lived greenhouse gases (GHG), alongside land-use change such as deforestation. It is a global challenge to mitigate climate change and therefore international efforts are required (IPCC et al., 2022), such as actions outlined in the 2015 Paris Agreement (UNFCCC, 2015) with a long term goal to limit global warming to 2 °C over pre-industrial levels. Recent UN Conferences of the Parties, i.e. COP26 in Glasgow, UK (UNFCCC, 2021) and COP27 in Sharm El-Sheikh, Egypt (UNFCCC, 2022b), requested Parties to revisit it and recognised the need to limit global warming to 1.5 °C over pre-industrial levels.

The UK has passed Net Zero emissions legislation, which requires the government to take actions to achieve net zero GHG emissions by 2050 (BEIS, 2019). Subsequently, the UK government set out its Net Zero Strategy to control GHG emissions (BEIS, 2021) in line with the 2015 the Paris Agreement. This strategy includes decarbonisation pathways (indicative delivery pathways by sectors, such as low carbon sources for electricity for power sector, use of zero emission vehicles for transport sector, and fuel switching, Carbon Capture, Usage & Storage for industry sector etc) with tracking for carbon budgets and Nationally Determined (emission reduction) Contributions. Decarbonisation measures aim to reduce or remove carbon emissions via a variety of climate actions and introduction/growth of low carbon technologies across several sectors, e.g. energy, industry, transport and residential (Garvey et al., 2022). Net Zero means that any inevitable carbon emissions (such as may arise from aviation and industry sectors) will be balanced by carbon removals, e.g. tree planting and the use of carbon capture and storage technology. The European Commission has also proposed the first European Climate Law

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in 2020 to legislate for a climate-neutral target by 2050 (EC, 2020). China has committed to Nationally Determined Contributions in 2015, strengthened in 2020 with the announcement of (targeted) peak carbon emissions in 2030 and achievement of carbon neutrality by 2060 (Shi et al., 2021a). India updated its Nationally Determined Contributions in 2022, and announced a target of achieving Net Zero by 2070 (UNFCCC, 2022a).

Air pollution (driven by toxic gaseous air pollutants such as nitrogen dioxide gas (NO_2) and particulate matter such as $PM_{2.5}$) is the biggest environment threat to human health, in terms of present day mortality drivers, worldwide. The World Health Organisation (WHO) estimated 4.2 million annual premature deaths occur globally due to outdoor air pollution (WHO, 2022). In the UK, long-term exposure to air pollution was estimated to cause 29–43,000 premature deaths each year or up to a sixth month reduction in life expectancy along with multiple adverse health outcomes and direct and indirect economic costs (COMEAP, 2018; Mannucci et al., 2019). The WHO updated its Global Air Quality Guidelines (AQGs, together with interim targets) in September 2021 (WHO, 2021), with lowered guidelines reflecting better understanding and stronger evidence of health risks at lower levels of air pollution compared with the previous guidelines published in 2005 (WHO, 2005). The WHO AQG represent the lowest level at which health effects are manifest, but there is no known lower threshold from a health perspective, and thus attempts to reduce air quality levels as much as possible should be made. This brings greater focus to reducing local, national and international emissions of air pollutants. It is estimated that 99% of the world's population lives in areas exceeding at least one of the 2021 WHO air quality guidelines (WHO, 2022). In the UK, the 2019 Clean Air Strategy aims to reduce by 50% the number of people living in locations with $PM_{2.5}$ concentrations greater than 10 μg m^{-3} (Defra, 2019b). ApSimon et al. (2021) investigated the drivers of $PM_{2.5}$ exceedances beyond 10 μg m^{-3} using the UK Integrated Assessment Model (UKIAM) and proposed useful indicators (population weighted mean concentration and population weighted mean exceedance) in setting targets and assessing policy impacts. The UK Department for Environment, Food & Rural Affairs (Defra) has recently released a range of Environmental Targets under the Environment Act 2021 (UK_PGA, 2023), including for air quality, setting both an annual mean concentration target (10 μg m^{-3} or below, to be achieved by 2040) and a population exposure reduction target (35% by 2040 compared to 2018) for $PM_{2.5}$ in England (drawing on ApSimon et al. (2023)). Previous NO_2 objectives (including 40 μg m^{-3} annual mean) remain in force.

Climate change and air quality are distinct but closely interlinked phenomena (Hassan et al., 2016; Hong et al., 2019; Thambiran and Diab, 2011). Carbon dioxide and short-lived air pollutants are often co-emitted during combustion processes of fossil fuels (Monks et al., 2009). Decarbonisation can therefore not only reduce carbon, but also air pollutant emissions (such as NO_2 and $PM_{2.5}$) (Chang et al., 2017). The spatial scale of the impacts differ: the long lifetime of most greenhouses gases (such as CO_2: decades) dictates global mixing and effects, while the much shorter atmospheric lifetimes of air pollutants (NO_2: 12–24 h; $PM_{2.5}$: days) give rise to much more localised effects – or, much more localised benefits from emission reductions. Climate mitigation policies targeting Net Zero emissions will likely bring regional and local improvement in air quality and associated health benefits (Wang et al., 2020; West et al., 2013). Dean and Green (2018) reviewed the impacts of climate change on air quality and associated human health in Sydney, Australia and suggested a coordinated way to tackle the challenges of climate change and air quality. Xia et al. (2015) assessed the impacts of mitigation policies in transport sector for Adelaide, Australia and found a reduction of 0.59 μg m^{-3} in $PM_{2.5}$ concentration for the scenario of 40% reduction in vehicle kilometres travelled. Agbossou et al. (2022) performed integrated assessment of priority climate change mitigation measures in reducing air pollution for Togolese in west Africa. Li et al. (2018) investigated air quality co-benefit of illustrative climate policies on carbon pricing in China. Zhu et al. (2022) investigated two long-term decarbonisation policies on building and truck electrification for California in the USA, and found air quality reduction of 0.68 μg m^{-3} and 0.59 μg m^{-3} in $PM_{2.5}$ exposure, respectively. Williams (2007) concluded that there are significant synergies between climate change and air quality policies in the UK and thus a concerted consideration would be needed. Calatayud et al. (2023) found that there would be a significant reduction in modelled NO_2 concentrations of 34–55% for a future 70% vehicle electrification scenario, with very little reduction in modelled $PM_{2.5}$ concentrations of 1–4%. Mehlig et al. (2021) predicted that a road transport electrification scenario in the UK would bring 36% reduction in NO_x and 1% reduction in $PM_{2.5}$ from 2018 to 2030, using the UK Integrated Assessment Model.

Both the WHO air quality guidelines and national/international climate change targets are ambitious and challenging (Kinney, 2018; Maione et al., 2016; Watts et al., 2015). Regional net zero policies are often not developed to optimise improvements to wider health determinants. There are potential missing opportunities to maximise co-benefits to health, e.g., by improving air quality, and targeting such improvements at the most vulnerable populations or areas. The UK's Net Zero policy to mitigate the future impacts of climate change by reducing greenhouse gas emissions and fossil fuel combustion could deliver significant air quality co-benefits. However, the degree of air quality-mediated health benefit is dependent on the net zero policies implemented and their local spatial distribution. Policy choices could deliver similar climate benefits but influence air quality quite differently in terms of the scale of effect, pollutant species altered and local populations impacted (Klausbruckner et al., 2016; Melamed et al., 2016; Shindell et al., 2012). This creates uncertainty in quantifying the air quality mediated benefits to health from the transition to net zero. Furthermore, some climate actions (e.g. biofuel plantation and rural afforestation) could cause air quality disbenefits, whereas others could exacerbate air quality-related health inequality through the relocation of greenhouse gases and air pollution emissions to other locations (Gallagher and Holloway, 2020, Royal_Society, 2021). There are "unintended consequences", with negative impacts on health and equality within local, regional and national boundaries. The density, demography and health geography of the affected population are all relevant to determining potential health effects (Al-Kindi et al., 2020; Koman et al., 2018). Similarly, potential health effects may also not be equally shared across communities – currently air quality results in a significant environmental health inequality with the most disadvantaged communities often experiencing the worst pollution levels (Fairburn et al., 2019; Ferguson et al., 2021). These populations are potentially less likely to be able to make climate and air quality-improving or mitigating changes to their lifestyles (Fairburn et al., 2019; Ferguson et al., 2021). Consequently, net zero-driven changes to air quality could worsen or improve inequalities at a range of scales.

The purpose of this paper is to quantify air quality co-benefits of regional/local Net Zero policies at a high spatial resolution, and to assess their contribution to population attainment of 2021 WHO interim targets and air quality guideline levels, using the West Midlands, UK as a case study, and hence to provide a specific example of the linkage between climate change mitigation options and air quality.

2. Methods

2.1. *Air quality modelling approach*

The modelling area covers the West Midlands Combined Authority (WMCA) region (consisting of 7 local authorities, i.e., Birmingham, Coventry, Dudley, Sandwell, Solihull, Walsall, and Wolverhampton) in the UK. The WMCA has a diverse population of about 2.9 million and a geographical area of about 902 km^2. It is also a representative metropolitan area in the UK. Air pollution improvement has become an environmental policy priority in the region.

We applied the high resolution ADMS-Urban air quality model for the West Midlands (the baseline model configuration for the year of 2016 was described in Zhong et al., 2021) in this study. ADMS-Urban is an advanced air dispersion model based on Gaussian plume distribution (Owen et al., 2000; Hood et al., 2018), which uses two parameters (the boundary layer depth and the Monin–Obukhov length) to represent the atmospheric boundary layer properties. A physics-based approach is used by the model, and a variety of data inputs (e.g., emissions, background, meteorology, advanced canyon, and urban canopy) are required. The model can represent the effect of complex urban morphology on pollutant dispersion and can capture the street-scale resolution air quality in urban environments. It has an option for the Generic Reaction Set (GRS) chemistry scheme (Venkatram et al., 1994; CERC, 2020) to consider chemical reactions for key reactive pollutants.

Emission sources (including point, road and grid source types) have been pre-processed using the EMIT Atmospheric Emissions Inventory Toolkit (CERC, 2021) before their implementation into the ADMS-Urban model. Point sources were obtained via the UK National Atmospheric Emissions Inventory (NAEI) dataset (NAEI, 2022) with additional stack parameters derived from Birmingham City Council (BCC) Airviro model (Airviro, 2018). Road sources were calculated from the traffic activity data and emission factors with real-world adjustments (Hood et al., 2018) in EMIT. Traffic activity data for heavy and light vehicles over the West Midlands was based primary on the PRISM (PRISM, 2019) traffic model from Transport for West Midlands (TfWM), with additional road links for Birmingham City Centre from the SATURN (BCC, 2018) traffic model from BCC. Traffic activity data for buses over the West Midlands was estimated based on Remix bus timetable data (Remix, 2019). Traffic fleet composition information was derived from BCC's Automatic Number Plate Recognition (ANPR) data. Grid sources for 11 SNAP (Selected Nomenclature for Air Pollution) sectors (Zhong et al., 2021) were obtained from the UK NAEI dataset (NAEI, 2022). Time varying profiles for the total gridded emissions were estimated based on time-varying factors used in the EMEP model (Simpson et al., 2012), which covers hour of the day, day of the week, and month of the year factors by pollutant and SNAP sector. Diurnal profiles for road emissions were estimated based on the local traffic survey data (provided by Transport for West Midlands). Background data for the West Midlands was obtained from the Automatic Urban and Rural Network (AURN) (Defra, 2019a). Meteorological data for Birmingham Airport available from the Met Office MIDAS dataset (Met_Office, 2019) was used as an appropriate synoptic site for the West Midlands. The advanced street canyon (Hood et al., 2021) and urban canopy (Hood et al., 2014) module was applied to capture the effects of street canyon and urban canopy flow on dispersion.

This modelling was performed with the task farming option (Zhong et al., 2021) in the ADMS-Urban model using the University of Birmingham's HPC cluster. The 2016 baseline ADMS-Urban model for the West Midlands has been validated against hourly measurement data from 12 regional background stations, 19 local roadside sites and 1 airport site (See S1 for model evaluation statistics from Zhong et al., 2021). Hourly model outputs were generated at the street scale resolution (10 m × 10 m). Annual mean concentrations were then calculated and spatially averaged to ward level for comparison with the 2021 WHO interim targets and guidelines. Here, four modelling scenarios were constructed (Table S2), i.e. :

1) Case "2021 BAU" represents a business-as-usual scenario (BAU) for 2021. It was configured based on the 2016 baseline model (Zhong et al., 2021), with traffic fleet composition updated to 2021. Activity levels were not adjusted for the ongoing "rebound" from Covid-19 reductions, rather than reflecting true activity (traffic) levels in 2021.
2) Case "2030 BAU" represents the business-as-usual scenario for 2030, which accounts for anticipated emission reductions in line with the National Emission Reduction Commitments Directive (NECD) commitments (EEA, 2019) and actions proposed in the 2019 Clean Air Strategy. This includes for example ongoing renewal of the vehicle fleet (with EVs reaching percentages of 3.8% for cars, 0.0015% for heavy vehicles, buses and coaches), and agricultural actions to reduce reactive nitrogen emissions, but without the larger and faster shifts envisaged in the more recent Net Zero strategies. Emissions for the 2030 BAU case for all SNAP sectors were simply scaled from these used in the 2016 baseline model to 2030 (Table S3). Scaling factors for regional background for 2030 scenarios were estimated based on the Defra projected background maps for West Midlands between 2016 and 2030 (Table S4). The traffic fleet composition projected for 2030 in EMIT was used for explicit road emissions.
3) Case "2030 NZS" (Net Zero Scenario) takes into account further emission reductions, which were estimated in line with the UK Net Zero Strategy released in 2021. The changes in carbon emissions were used for the calculation of scaling factors between 2016 and 2030 for air pollutant emissions in the relevant SNAP sectors, alongside a judgement-based adjustment – reflecting the nature of the emitting activities – to capture the proportionality, or otherwise, between co-emissions of CO_2 and the relevant air pollutants (Table S3). Traffic fleet composition for 2030 (representing transport decarbonisation) set out in the UK Net Zero Strategy was used for explicit road emission calculations, i.e., with electric vehicles as a fleet percentage representing 24% of cars, 9% of heavy vehicles and 25% of buses and coaches (See the UK Net Zero Strategy (BEIS, 2021)).
4) Case "2030 EV" reflects a modification of the 2030 NZS scenario, in which we only consider transport decarbonisation actions, while emissions other SNAP sectors (for example industry, power generation) were held at their 2030 BAU levels. The traffic fleet composition used was identical to that in the 2030 NZS scenario.

2.2. Population attainment data analysis

Population data were derived from the Office for National Statistics (ONS) Demographic Brief. An estimate for the population in 2030 for the WMCA was generated. This utilised the 2018 Office for National Statistics census population figures (for the analysis for 2021 BAU) and estimated the amount of growth and total population in 2028 and 2038 for the seven local authorities and the 192 wards that make up the WMCA. To determine the 2030 population for each local authority the mean yearly increase between the 2028 and 2038 population estimates was calculated (population growth/10) and then added to the 2028 total (twice: 2028 to 2030). To apply this local authority population increase to wards, the growth for each local authority from 2018 to 2030 was applied to each constituent ward as a single scaling factor. This in effect assumes that there are no dramatic changes in relative population patterns within Local Authorities/between wards.

Index of multiple deprivation (IMD) scores for the WMCA wards were obtained from ONS. These scores were treated in deciles. We note that the IMD score includes an environmental component (9.3% of the IMD score), which in turn draws upon air quality (specifically, emissions of nitrogen dioxide, benzene, sulphur dioxide and particulates) - however this air quality component only makes up 1.5% of the total IMD score so the circularity risk is minimal (BROOK and KING, 2017; McLennan et al., 2019). The high resolution AMDS-Urban model was utilised to extract the spatially averaged NO_2 and $PM_{2.5}$ concentrations for each ward under each scenario within Esri Arcmap and then analysed within R Studio.

Prior to analysis, pre-test checks for normality, homogeneity and outliers were performed through skewness, kurtosis, Levene's and Shapiro-Wilks tests and generation of boxplots within R that identified data points outside of the interquartile range. Consequently, data was logged and outliers removed where necessary. Mean pollution concentrations for wards were compared to the 2021 World Health Organisation (WHO) Air Quality Guidelines and interim targets (AQGs – Table S5) to determine the number of wards exceeding the interim targets and guideline values. This also yielded an estimated population of the region exposed to exceeding pollution levels. Correlation and linear regression analysis was performed between IMD scores for each ward and the mean NO_2 and $PM_{2.5}$ values. One-way analysis of variance was also performed through Kruskal-Wallace and post-hoc Dunn Tests to determine whether there were significant differences between the mean pollution concentrations found in the 10 deciles of IMD scores.

3. Results

3.1. Changes in air quality over the West Midlands for 2021 and 2030 business-as-usual scenarios

The baseline ADMS-Urban model configuration for the West Midlands has been evaluated against observations with satisfactory agreement (Zhong et al., 2021). Fig. 1a and b shows predicted annual air quality maps of NO_2 at 10 m × 10 m resolution over the West Midlands for 2021 and 2030 BAU scenarios. The simulations show a fall in NO_2 concentrations of between 2021 BAU and 2030 BAU by up to 28 μg m^{-3} (or 37%). This reflects the impact of NECD commitments (EEA, 2019) and 2019 Clean Air Strategy on air quality, primarily the reduction in vehicle emissions from penetration of more modern (lower emission) vehicles, and expected growth in EVs. Patterns of the influence of road emissions on air pollution dispersion are clear, with relatively high levels of NO_2 close to highways and busier major roads towards the city centres. Due primarily to changing NO titration chemistry effects, there is an increase in O_3 levels (See Fig. S1) of up to 6 μg m^{-3} (or 23%) while NO_2 levels are predicted to reduce in cities, which is consistent with Sicard et al. (2023). Fig. 1c and d further shows predicted annual air quality maps of $PM_{2.5}$ for the 2021 BAU and 2030 BAU scenarios. There are significantly smaller changes (reductions) in $PM_{2.5}$ concentrations predicted between 2021 BAU and 2030 BAU scenarios of about 1 μg m^{-3} (or equivalent to 8%). These reductions in NO_2 and $PM_{2.5}$ are consistent with Mehlig et al. (2021), taking into account the effect of road transport electrification.

3.2. Assessing air quality cobenefits of net zero policies

Fig. 2 shows the impacts of 2030 Net Zero and road transport decarbonisation policies (UK, 2021) on NO_2 concentrations over the West Midlands compared with the 2030 BAU scenario. The model predictions indicate that there could be a reduction in annual mean NO_2 of up to about 6 μg m^{-3} (or 21%), relative to the NO_2 concentrations expected for 2030 without Net Zero measures (i.e., relative to 2030 BAU). Fig. 2a captures the changes in NO_2 concentration at the regions both near and away from major roads, which to some extent reflect the effect of decarbonisation in the road transport sector (indicated in Fig. 2b with near road dispersion pattern) and other sectors (such as power generation, industry, fuel supply and hydrogen, heat and buildings etc), respectively.

The model results indicate a smaller reduction in annual mean $PM_{2.5}$ than for NO_2, of up to 1.4 μg m^{-3} (or 12%) (Fig. 3), due to 2030 Net Zero policies (i.e. 2030 NZS vs 2030 BAU). There are no clear near-road patterns in PM change, and the effect of the additional electrification

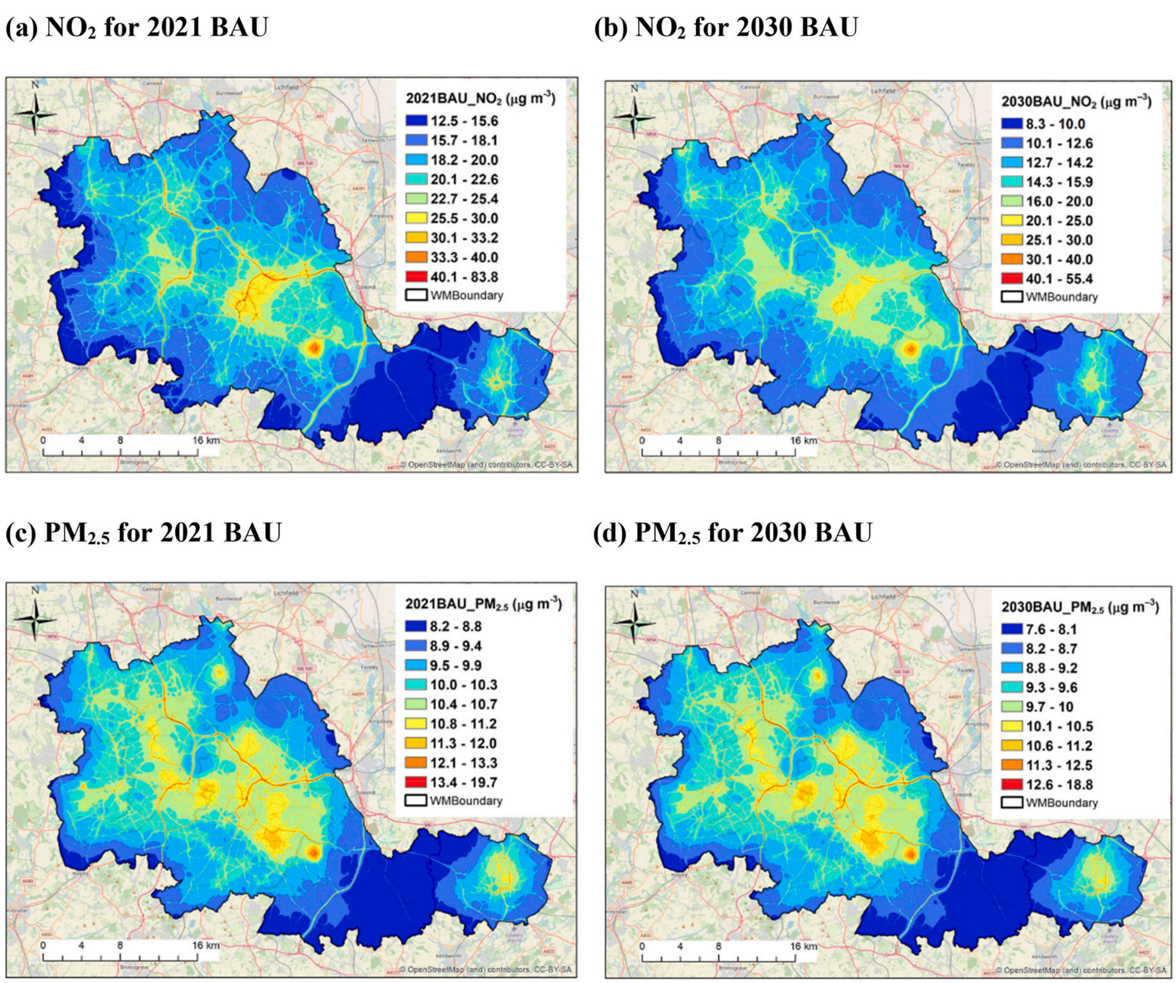

Fig. 1. Air quality maps of annual mean concentrations of NO_2 (a, b) and $PM_{2.5}$ (c, d) (in μg m^{-3}) at 10 m × 10 m resolution for 2021 BAU and 2030 BAU scenarios. BAU represents the business-as-usual scenario.

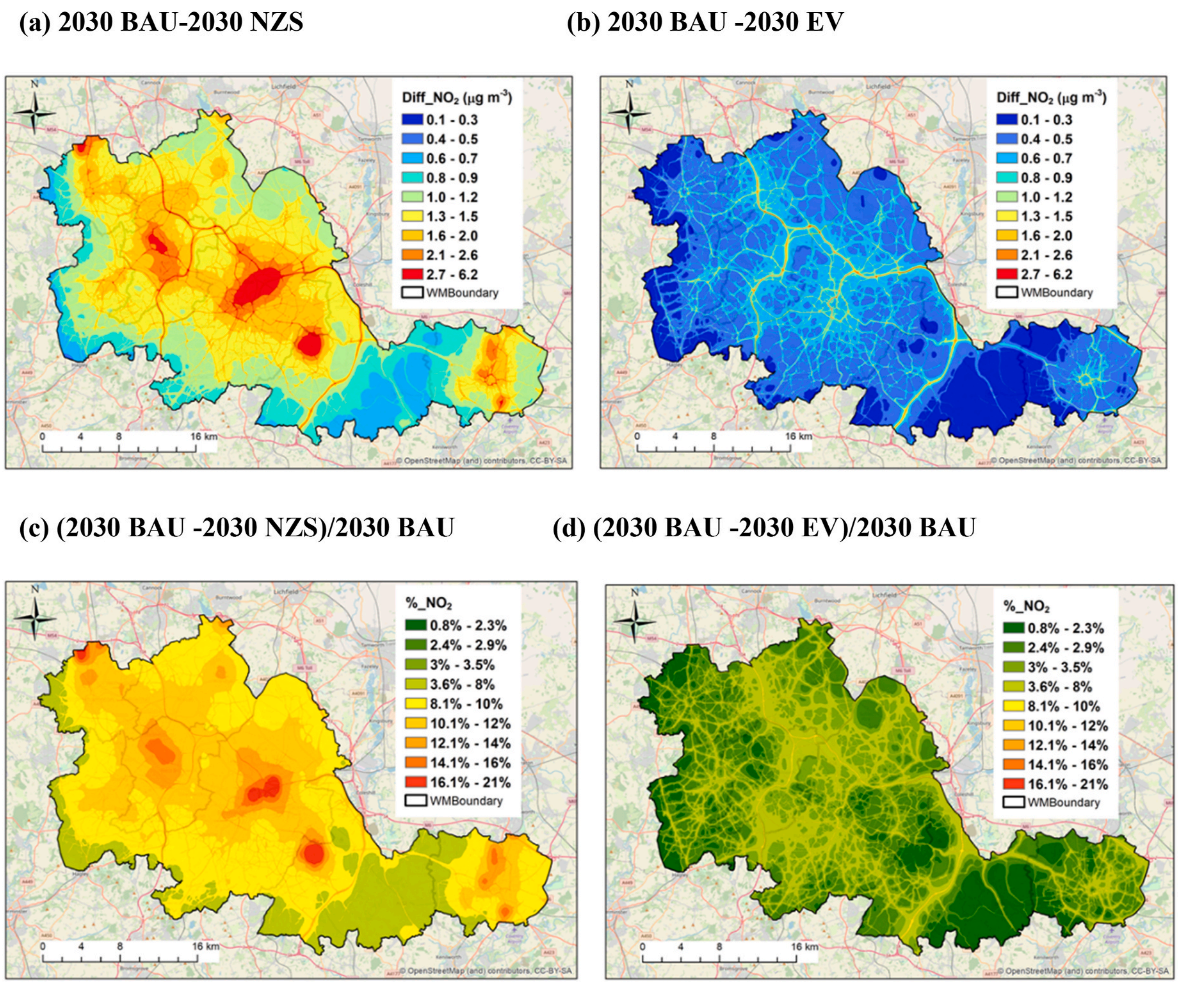

Fig. 2. Reduction in annual mean NO_2 (in μg m^{-3}) for (a) 2030 NZS and (b) 2030 EV scenario relative to 2030 BAU scenario; and percentage change for (c) 2030 NZS and (d) 2030 EV scenario relative to 2030 BAU scenario. BAU represents the business-as-usual scenario. NZS represents the Net Zero Scenario. EV represents the electric vehicles (transport decarbonisation) scenario.

in the transport sector (electrification only with the assumption of the same traffic activity levels) on $PM_{2.5}$ is almost negligible (Fig. 3b, d). Although the fleet changes to EVs were captured, non-exhaust $PM_{2.5}$ emissions remain (in both cases). The impacts of other emission sectors are indicated in Fig. 3a, c, reflecting relatively larger reductions for $PM_{2.5}$ concentrations in urban areas compared with rural areas.

3.3. Population attainment of 2021 WHO interim targets and air quality guideline levels

Table 1 shows the number of wards and the corresponding estimates of the regional population exposed to air pollution with annual mean concentrations exceeding WHO interim targets three and four (IT3/IT4, shown in Table S5) and the AQG (WHO, 2021) for NO_2 and $PM_{2.5}$. For the 2021 BAU scenario, there are 78 wards exceeding IT3 for NO_2. IT3 is only exceeded by three and two wards respectively under the 2030 BAU and 2030 EV scenarios for NO_2 (less than 2% of the population) with no wards exceeding the limit value under the 2030 NZS scenario. There are no wards where the ward-level average concentration of NO_2 exceeds 30 μg m^{-3} (IT2) under any of the scenarios, and IT4 is not available for NO_2 in the WHO AQG. However, a large proportion of wards exceed the actual AQG for NO_2 under all scenarios, with 92–99% of the population being exposed to ward-level annual mean levels over 10 μg m^{-3}.

For $PM_{2.5}$, no wards exceed IT3 under any of the scenarios. For IT4, there are 78 wards exceeding 10 μg m^{-3} under the 2021 BAU scenario. No wards exceed IT4 under the 2030 NZS scenario but 28 and 26 wards show average annual mean concentrations exceeding IT4 for the 2030 BAU and 2030 EV respectively, which represent 16% and 14% of the region's population. Finally, the simulated ward average annual mean $PM_{2.5}$ concentration exceeds the $PM_{2.5}$ AQG of 5 μg m^{-3} in all scenarios, in both 2021 and 2030, across the region. This is consistent with attainment for 2040 predicted for the UK by Oxley et al. (2023) using the UK Integrated Assessment Model and EMEP4UK models. The lower bound to $PM_{2.5}$ levels projected is influenced by the unavoidable emissions from natural background $PM_{2.5}$ (such as sea salt) and transboundary (and shipping) contributions (Oxley et al., 2023).

Table 2 shows annual mean concentrations of NO_2 and $PM_{2.5}$ by local authority and for the West Midlands overall (calculated based on the 10 m × 10 m air quality maps which were averaged over regional boundary lines)under each scenario. There are two local authorities (Birmingham and Sandwell) with average NO_2 concentrations exceeding IT3 (20 μg m^{-3}) for the 2021 BAU scenario. All 2030 scenarios predict NO_2 concentrations lower than IT3 for all local authorities. For all scenarios, all local authorities exceed the WHO AQG (10 μg m^{-3}), in both 2021 and 2030. For $PM_{2.5}$, no local authorities exceed IT3 (15 μg m^{-3}), but all would exceed the WHO AQG (5 μg m^{-3}). Under the 2021 BAU scenario, the model predicts that two local authorities (Birmingham and Sandwell) have average $PM_{2.5}$ concentrations which exceed IT4 (10 μg m^{-3}). All 2030 scenarios predict $PM_{2.5}$ concentrations lower than IT4 for all local authorities.

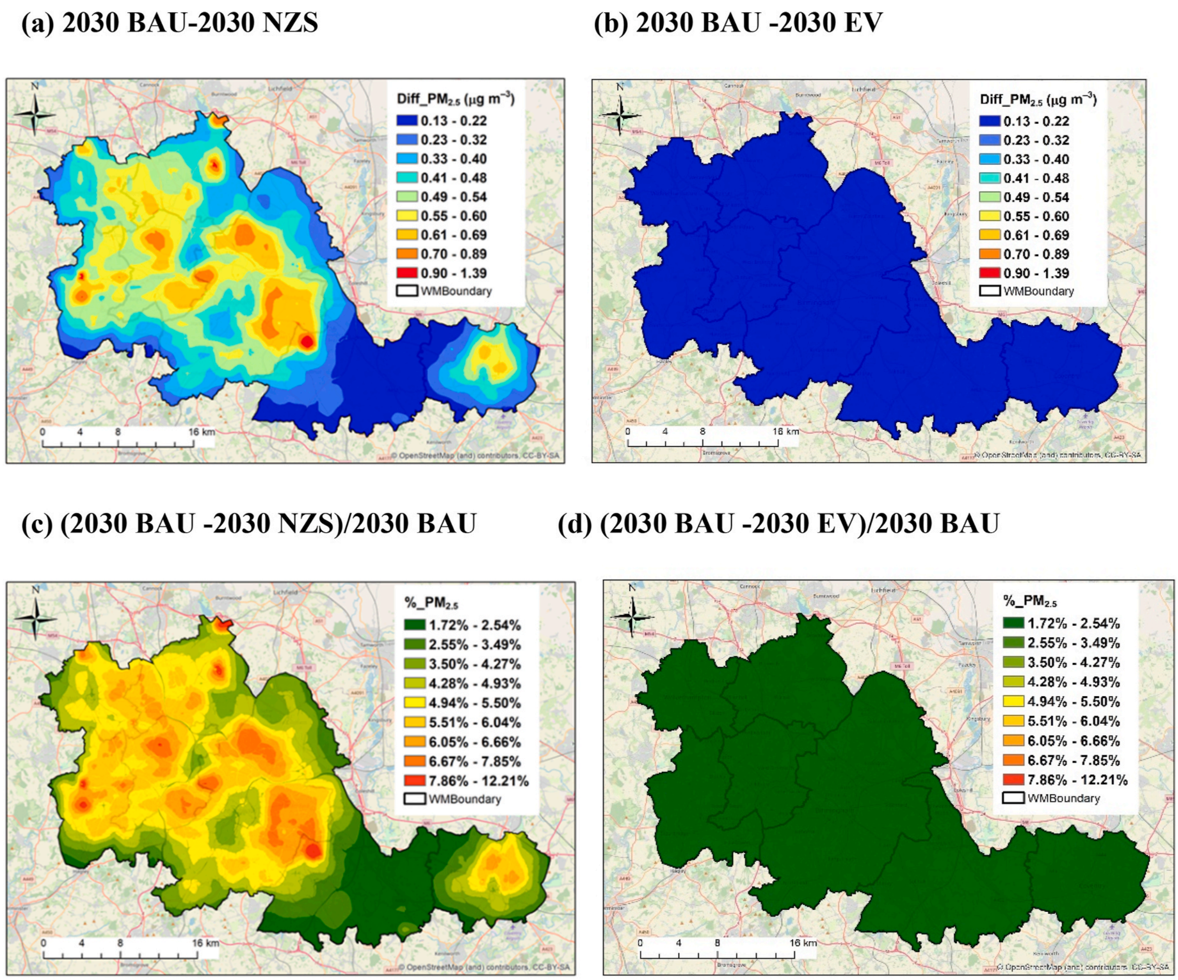

Fig. 3. Reduction in annual mean $PM_{2.5}$ (in μg m^{-3}) for (a) 2030 NZS and (b) 2030 EV scenario relative to 2030 BAU scenario; and percentage change for (c) 2030 NZS and (d) 2030 EV scenario relative to 2030 BAU scenario. BAU represents the business-as-usual scenario. NZS represents the Net Zero Scenario. EV represents the electric vehicles (transport decarbonisation) scenario.

Table 1
The number of wards and respective population in the WMCA exceeding IT3/4 and the AQG for NO_2 and $PM_{2.5}$ under each scenario.

Pollutant	Scenario	Wards Exceeding IT3 (20 μg m^{-3})	~Population (%)	# Wards Exceeding IT4 (NA)	~Population (%)	Wards Exceeding AQG (10 μg m^{-3})	~Population (%)
NO_2	**2021 BAU**	78	1.30 m (44.57%)	NA	NA	192	2.92 m (100%)
	2030 BAU	3	0.047 m (1.53%)	NA	NA	189	3.08 m (98.79%)
	2030 EV	2	0.032 m (1.04%)	NA	NA	187	3.05 m (97.97%)
	2030 NZS	0	0 (0)	NA	NA	174	2.87 m (92.07%)
Pollutant	**Scenario**	**Wards Exceeding IT3 (15 μg m^{-3})**	**~Population (%)**	**Wards Exceeding IT4 (10 μg m^{-3})**	**~Population (%)**	**Wards Exceeding AQG (5 μg m^{-3})**	**~Population (%)**
$PM_{2.5}$	**2021 BAU**	0	0 (0)	104	1,665,697 (57.11)	192	2.92 m (100%)
	2030 BAU	0	0 (0)	28	494,121 (15.87)	192	3.11 m (100%)
	2030 EV	0	0 (0)	26	458,441 (14.73)	192	3.11 m (100%)
	2030 NZS	0	0 (0)	0	0 (0)	192	3.11 m (100%)

3.4. Pollution concentrations and deprivation

Impacts on environmental health inequality were assessed by evaluating the relationship between ward-level average NO_2 and $PM_{2.5}$ concentrations, and the ONS Index of Multiple Deprivation (IMD) decile, under baseline and 2030 Net Zero scenarios. Regression analysis of NO_2 and $PM_{2.5}$ (Fig. 4) concentrations exhibited a significant positive correlation ($p=<0.01$) with IMD. Furthermore, Kruskal-Wallace and post-hoc Dunn Tests showed statistically significant differences ($p=<0.05$) between the least deprived wards and the most deprived.

There was a slight reduction in the coefficient (slope) between the IMD score, and both $PM_{2.5}$ and NO_2, with both the gradient and absolute

Table 2
Local authority and West Midland regional averaged concentrations of NO_2 and $PM_{2.5}$ under each scenario.

Pollutants	NO_2 (μg m^{-3})				
Scenario	2021 BAU	2030 BAU	2030 EV	2030 NZS	Estimated 2030 population (million)
Birmingham	20.41	14.52	14.00	12.98	1.19
Coventry	17.68	12.36	11.96	11.14	0.418
Dudley	17.29	12.02	11.62	10.91	0.337
Sandwell	21.22	15.21	14.65	13.48	0.348
Solihull	17.19	12.02	11.66	10.98	0.230
Walsall	19.29	13.61	13.10	12.21	0.304
Wolverhampton	18.22	12.94	12.55	11.59	0.280
West Midlands	18.92	13.36	12.90	12.01	3.11
IT3	20	20	20	20	
IT4	NA	NA	NA	NA	
AQG	10	10	10	10	
Pollutants	$PM_{2.5}$ (μg m^{-3})				
Scenario	2021 BAU	2030 BAU	2030 EV	2030 NZS	
Birmingham	10.10	9.43	9.42	8.94	
Coventry	9.44	8.78	8.77	8.42	
Dudley	9.73	9.08	9.07	8.62	
Sandwell	10.44	9.76	9.75	9.20	
Solihull	8.88	8.23	8.23	7.98	
Walsall	9.97	9.31	9.31	8.82	
Wolverhampton	9.86	9.18	9.18	8.71	
West Midlands	9.74	9.08	9.07	8.65	
IT3	15	15	15	15	
IT4	10	10	10	10	
AQG	5	5	5	5	

difference in concentration between the most and least deprived wards reducing under the 2030 NZS scenario, relative to the 2021 BAU scenario (difference in mean concentration, between decile 1 and decile 10, reduced from 4.1 to 3.62 μg m^{-3} for NO_2; 1.42 to 1.17 μg m^{-3} for $PM_{2.5}$). A (small) reduction in air pollution inequality – as assessed by the relationship between annual mean concentrations and IMD – is predicted to arise from the Net Zero policy measures. In contrast, under the EV-only scenario (2030 EV), the air pollution inequality for NO_2 decreases, but that for $PM_{2.5}$ increases – with the difference in annual mean concentrations of $PM_{2.5}$ for the first and last IMD deciles increasing, for 2030 EV relative to 2021 BAU (Table 3). This does assume no change in IMD distribution over the scenario period.

4. Discussion

All simulations for 2030 show improvements in air quality, relative to the 2021 simulation, reflecting reduced within-region emissions and corresponding background pollutant reductions. For NO_2, almost all ward areas in the West Midlands will meet the WHO interim target (IT) 3 for NO_2 (20 μg m^{-3}) for the 2030 BAU and 2030 EV scenarios, and all meet this threshold for the 2030 NZ scenario. However, only a minority of ward-average annual mean NO_2 concentrations comply with the WHO air quality guideline for NO_2 of 10 μg m^{-3} across the scenarios considered – for the greatest emission reduction, the 2030 NZS scenario, 174 out of 192 wards, representing 92% of the region's population, remain above this level. Net Zero policy measures, as approximated here, will improve air quality across the region, but are not sufficient to reach the WHO guideline levels, i.e. health impacts from NO_2 exposure will remain.

Decarbonisation in transport sectors will drive the reduction in NO_2 concentrations close to road sources, while decarbonisation in other sectors will bring further reduction especially areas far away road sources. Nonetheless, very substantial local and regional reductions in NO_2, from 18.9 to 12.0 μg m^{-3} (averaged across the region: Table 2) are achieved. The greatest relative reduction in local authority (LA) averaged annual mean NO_2 concentrations from implementation of Net Zero policies is predicted for Sandwell, reflecting the spatial distribution of

Table 3
Difference between the most and least deprived ward air pollution levels (μg m^{-3}) under the BAU, EV and NZS scenarios.

	Difference (most – least deprived)			
	2021 BAU	2030 BAU	2030 EV	2030 NZS
NO_2	5.56 (IMD 1 ± 1.8, IMD 10 ± 2.5)	4.1 (IMD 1 ± 1.6, IMD 10 ± 1.87)	4.3 (IMD 1 ± 1.32, IMD 10 ± 1.8)	3.62 (IMD 1 ± 1.29, IMD 10 ± 1.23)
$PM_{2.5}$	1.39 (IMD 1 ± 0.32, IMD 10 ± 0.38)	1.42 (IMD 1 ± 0.42, IMD 10 ± 0.3)	1.42 (IMD 1 ± 0.42, IMD 10 ± 0.48)	1.17 (IMD 1 ± 0.33, IMD 10 ± 0.46)

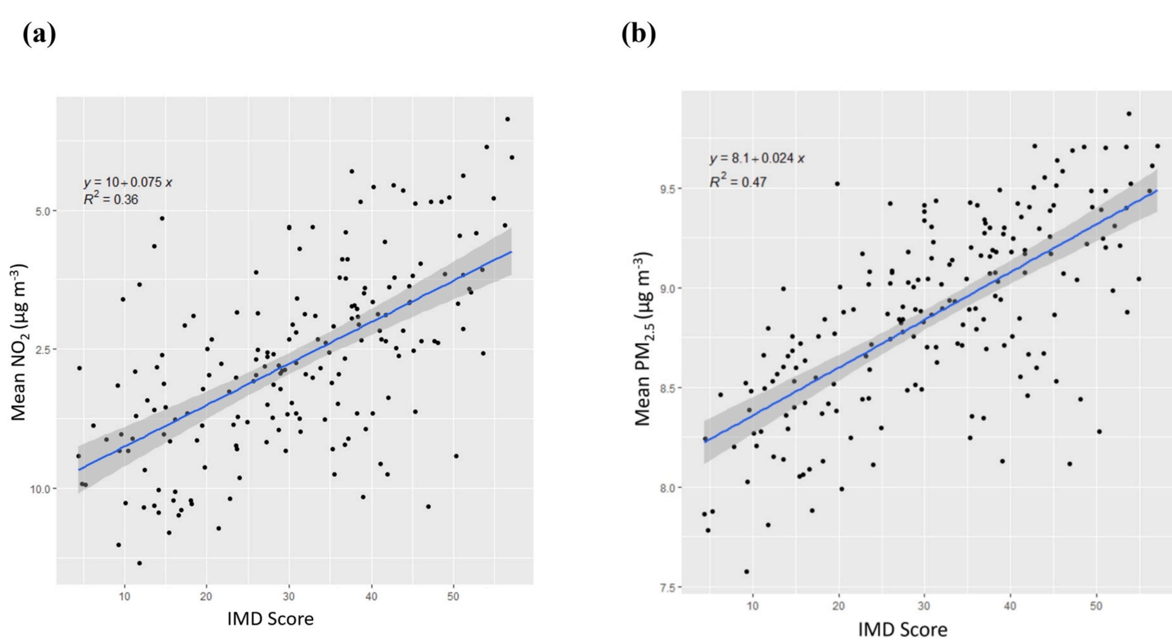

Fig. 4. Relationship between (a) mean NO_2 and IMD (Index of Multiple Deprivation) score and (b) mean $PM_{2.5}$ and IMD score for the 2030 NZS scenario.

modelled emission sources, although all LAs show similar fractional reduction in NO_2. Our results indicate that further clean air actions are needed to reduce annual average NO_2 concentrations to meet the WHO AQG for NO_2. Although Net Zero policies will contribute to a reduction in NO_x emissions, leading to lower (secondary) NO_2 concentrations in the West Midlands (Fig. 2) and potentially secondary nitrate in fine particles (Mazzeo et al., 2022), but it will result in a significant increase in O_3 concentrations (Fig. S2) due to a weakened titration by NO (Wang et al., 2022; Chameides et al., 1997; Clapp and Jenkin, 2001). This result is consistent with recent findings for cities in China (Liu et al., 2023) and globally (Sicard et al., 2023; Shi et al., 2021b).

For $PM_{2.5}$, the number of wards for which the ward-averaged annual mean $PM_{2.5}$ concentration exceeds the WHO interim target (IT) 4 for $PM_{2.5}$, of 10 μg m^{-3} is 28 and 26 wards out of 192, representing 16 and 15% of the region's population, for the 2030 BAU and 2030 EV scenarios respectively. All ward-average annual mean $PM_{2.5}$ levels comply with IT4/the UK 2030 target, 10 μg m^{-3}, under the 2030 NZS scenario – demonstrating the direct regional air quality and hence regional health co-benefit of net zero policies. However, no wards achieve compliance with the WHO air quality guideline for $PM_{2.5}$ (5 μg m^{-3}), under any of the scenarios, i.e. health impacts from $PM_{2.5}$ exposure will persist, consistent with Oxley et al. (2013). While ward-average annual mean $PM_{2.5}$ levels may sit below the 10 μg m^{-3} target in some 2030 scenarios, localised areas where $PM_{2.5}$ levels exceed this level will remain (see Fig. 1d), consistent with the findings of ApSimon et al. (2023). Mazzeo et al. (2022) showed most of the $PM_{2.5}$ was from regional sources for the West Midlands and regional mitigation policies to reduce domestic combustion is the most effective policy. Further regional and national efforts will be needed to reduce $PM_{2.5}$ concentrations. Recently, agriculture has been highlighted as an important transboundary source for $PM_{2.5}$ (Kelly et al., 2023; Wyer et al., 2022; Ma et al., 2021), with a larger influence than city sources (such as the large urban areas in the West Midlands). Transport decarbonisation (vehicle electrification) would bring significant reduction in NO_2 concentrations, but a much smaller effect on $PM_{2.5}$, consistent with Calatayud et al. (2023). This is because exhaust particle emissions are relatively low in the UK now and primary non-exhaust $PM_{2.5}$ emissions are unlikely to reduce (Defra, 2023).

The air quality inequality observed under the present day (baseline scenario) – where those wards which are the most deprived suffer the worst air quality (Fig. 4) (also found byBrunt et al., 2016)– is reduced under the 2030 NZS scenario, with both the absolute difference in NO_2 and $PM_{2.5}$ levels between the most and least deprived deciles falling (Table 3). A greater reduction is obtained for the Net Zero (2030 NZS) scenario, than for the 2030 BAU scenario. However, the air quality inequality for NO_2 (the difference in NO_2 levels predicted for the most vs least deprived wards) worsens for the 2030 EV scenario, relative to the business-as-usual (2030 BAU) case. The inference is that road-transport related climate policies alone – such as encouraging fleet shift to electric vehicles without wider policy measures – will not alleviate environmental health inequalities as much as implementation of a comprehensive Net Zero policy portfolio. Analysis suggests that the most deprived areas of the region are exposed to the highest air pollution levels, as has been shown in previous work and reviews (Fairburn et al., 2019). Future policies should pay more attention to the most deprived areas to reduce inequality.

5. Uncertainties and limitations

Our modelling is based on approximations, which may contribute to uncertainties in our results. These must be borne in mind when interpreting the results. The future scenario development here is driven by the Committee on Climate Change (CCC) 6th carbon budget assumptions, and an estimation was made in translating these changes in carbon emissions to their air pollutant equivalents. It is not driven by specific technology development trajectories/timelines. We recognize that the future emission scenarios approach used here, and in particular estimates of the air quality emission changes alongside a given/anticipated carbon emission shift, are uncertain. Emissions for the 2030 scenarios are estimated based on scaling factors by SNAP sector, which may not reflect specific abatement measures or strategies, or technological developments. The approach of scaling factors for future scenarios was also used in Kitwiroon et al. (2019). Future work to develop air pollutant co-emission profiles for specific technology, activity scenarios is a key requirement to reduce this uncertainty. The 2021 BAU scenario did not consider Covid impacts so the true activity (traffic) levels in the unusual year of 2021 were not reflected by the model, i.e. the 2021 BAU was a simulated scenario which did not reflect the real situation in 2021. It is also noted that the Euro 6 vehicle standard covers a wide range of technology and latest derived standards (Euro 6b, 6c and 6d). Euro 6 d, if delivered under real-world conditions and significant fleet penetration, would give a significant reduction in NO_x emissions for 2030, beyond that considered here. Similar assumptions apply to the background levels implemented within the model, which may not accurately reflect the background encountered in 2030. Meteorological conditions were not varied between the 2021 BAU and 2030 scenarios, while in practice the climatology is expected to change, with (broadly) windier wetter winters reducing the likelihood of high pollution winter stagnation events, and drier summers with more frequent secondary pollution episodes influenced by emissions from continental Europe (Royal_Society, 2021). Spatial averaging of results over ward and LA geographies will greatly smooth the variability actually present (Fig. 1), averaging out both the lowest and highest concentrations/hot spots at which significantly greater exposure, in excess of legal limits, may occur. While we recognize the uncertainties discussed above, the overall trend in air quality changes should be similar and thus our overall conclusions apply.

Future work should improve the emission projections, as sector knowledge improves, and use such new data and modelling to identify and prioritise Net Zero policies for reducing emissions from the sectors with maximum local air quality, health and economic benefits (or minimum disbenefits). Such data will help demonstrate that regional and local decarbonisation actions to tackle the global climate challenge deliver regional and local population co-benefits, mediated through improved air quality and hence health.

6. Conclusions

High resolution air quality modelling was conducted for a range of future business as usual and Net Zero scenarios over the West Midlands using the ADMS-Urban model. Air quality co-benefits of Net Zero scenarios for the West Midlands have been quantified. Net Zero scenarios will significantly improve air quality in the West Midlands, particularly for NO_2 (mostly through the electrification of the vehicle fleet) with reductions of up to 6 μg m^{-3} (or 21%) predicted for 2030, relative to levels expected under the "business as usual" approach. This demonstrates the very significant local and regional air quality – and hence health – co-benefit achievable from carbon emission reductions motivated by the global climate challenge. The improvements for $PM_{2.5}$ are more modest if only transport electrification (EV-only scenario) is considered, since non-exhaust road traffic emissions still remain. Net Zero scenarios with emission changes in all sectors (such as power generation, industry, fuel supply, heat and building etc) would achieve a reduction in annual mean $PM_{2.5}$ of up to 1.4 μg m^{-3} (or 12%). Net Zero scenarios will slightly reduce environmental health inequality, narrowing the air pollutant concentration gap between the most and least deprived wards for NO_2. The BAU scenario brings most wards below $PM_{2.5}$ concentrations of 10 μg m^{-3} (15% population in wards above this level) whilst the Net Zero scenario brings all ward-annual-mean $PM_{2.5}$ levels below 10 μg m^{-3}, meeting the UK $PM_{2.5}$ target a decade early, in this region. However, no ward-level-averaged annual mean $PM_{2.5}$ concentrations meet the WHO Air Quality guideline level of 5 μg m^{-3} under any scenario; similarly for NO_2 only 18 wards (8% of the region's population) have predicted NO_2 concentrations below the WHO guideline level (10 μg m^{-3}). Decarbonisation policies linked to Net Zero deliver substantial regional air quality benefits, but are not in isolation sufficient to deliver clean air with pollutant levels low enough to remove impacts upon health. The "EV scenario" delivers significantly smaller $PM_{2.5}$ benefits because of the unavoidable (i.e., maintained) non-exhaust emissions, and a slight inequality disbenefit, compared with the full Net Zero scenario. This study emphasises the policy importance of looking beyond road traffic (significantly reducing exhaust NO_x emissions) in maximising air quality-climate co-benefits.

Credit author statement

JZ: Methodology, Software, Visualization, Formal analysis, Writing – original draft. **JH:** Methodology, Visualization, Formal analysis, Writing – original draft. **WJB**: Conceptualization, Supervision, Writing – review & editing, Funding acquisition. **ZS**: Writing – review & editing, Project administration, Funding acquisition.

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Declaration of competing interest

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

Data availability

Data will be made available on request.

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Appendix A. Supplementary data

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