



Full length article

Climate change policies reduce air pollution and increase physical activity: Benefits, costs, inequalities, and indoor exposures

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ABSTRACT

The burden of diseases attributable to air pollution is comparable to those of global health risks such as unhealthy diets and tobacco smoking, with many air pollution sources also emitting climate heating gases. In this UK study we estimated the co-benefits of Net Zero (NZ) climate policy on the health benefits of air pollution reduction, increased active travel, outdoor exposure inequalities and indoor air pollution changes. The study focused on two of the largest UK sources, road transport and building heating, with comparisons made between NZ and UK existing policy, referred to as Business as Usual (BAU). Particulate matter (PM_{2.5}), Nitrogen Dioxide (NO₂) and Ozone (O₃) projections were made between 2019 and 2050, with emphasis placed upon the NZ co-benefits in 2030 and 2040. We compared the UK BAU scenarios with the Climate Change Committee's (CCC) Balanced Net Zero Pathway (BNZP) and Widespread Innovation (WI) pathway.

Compared to BAU predictions, BNZP assumptions lead to more electric vehicles, reduced vehicle km, more low carbon building heating, and reduced emissions of NO₂ and PM_{2.5}. By 2040 under BNZP, relative to BAU, the buildings sector was predicted to be three times more effective at reducing PM_{2.5} than road transport. To help reduce the inequality gap the NZ building transition was tailored toward those most in need. Outdoor air pollution exposure inequalities prevailed across the socioeconomic spectrum, especially for NO₂, but were less pronounced due in part to NZ policies. Core air quality health benefits for the BNZP buildings sector were £21.3 billion (16.4 to 26.2) by 2050 and £98.4 billion (75.7 to 121.1) by 2154. For the transport sector the health benefits were £9.1 billion (7.0 to 11.2) by 2050 and £36.5 billion (28.1 to 44.9) by 2154. NZ building sector operating costs did not achieve break-even via efficiency savings, but with Greenhouse Gas (GHG) (lower benefits) break-even was achieved in 2052. With additional air pollution health benefits, building-sector time to break-even improved by between 3.1 (2.5 to 4.7) and 6.3 (4.7 to 7.6) years to between 2046 and 2049. Analysis found that removing gas cooking at home, for NZ, may result in greater concentration reductions than outdoor air pollution for NO₂. Net Zero health and economic co-benefits are large, as are the changes needed, requiring political leadership and public engagement. Our findings are relevant to other countries facing the NZ transition.

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1. Introduction

The burden of disease attributable to air pollution is comparable to that of global health risks such as unhealthy diets and tobacco smoking, with evidence of increasing health impacts (WHO Global air quality guideline, 2021) and with many air pollution sources also emitting climate heating gases. Air pollution is considered the top environmental risk to human health in the UK, with 2019 PM_{2.5} and NO₂ Population Weighted Average Concentrations (PWAC) used by Mitsakou et al., (Mitsakou et al., 2022) to calculate the mortality burden. UK PWAC for PM_{2.5} was 9.0 µg m⁻³, ranging from 9.6 µg m⁻³ in England to 5.5 µg m⁻³ in Scotland, and for UK NO₂ was 14.2 µg m⁻³, ranging from 15.1 µg m⁻³ in England to 8.4 µg m⁻³ in Northern Ireland. Using updated concentration response functions from the Committee on the Medical Effects of Air Pollutants (COMEAP), (Committee on the Medical Effects of Air Pollutants) Mitsakou concluded that the long-term exposure to air pollution in 2019 contributed a mortality burden equivalent to 29,000–43,000 premature deaths. (Mitsakou et al., 2022) Air pollution health impact assessments are well established in the UK for mortality and hospital admissions, with recently determined concentration–response functions (CRFs) for a range of morbidity outcomes now available and based on systematic reviews of exposure to PM_{2.5}, NO₂ and O₃ and disease incidence. (Forastiere et al., 2024) Whilst there is compelling evidence to support the health effects of outdoor air pollution, reflected in the joint European Respiratory Society/American Thoracic Society policy statement, (Thurston et al., 2017) there is also a need to address indoor air pollution, (Chief Medical Officer's Annual Report) and, exemplified by the IPCC Synthesis Report, (AR6 Synthesis Report: Climate Change, 2023) for people and Governments everywhere to commit to climate action.

The UK has adopted legally binding targets for Net Zero (NZ) (UK Government Net Zero Law) that are consistent with the Paris agreement, (Schleussner et al., September 2016) and has independent advice on achieving NZ goals from the Climate Change Committee (CCC). Therefore, UK policy is being developed to meet NZ by 2050, but without full consideration of the benefits/dis-benefits on human health from air pollution both indoors and outdoors. It is possible for NZ policy to help air pollution in simultaneously reducing emissions from common sources, improve people's health, (Milner et al., 2023; Williams et al., 2018) improve resilience by reducing the burden on health systems, benefit the country economically (Karlsson et al., 2020) and reduce exposure inequalities. However, it could also present a burden by for example, increasing indoor air pollution through making homes more airtight, (Wilkinson et al., 2009) increase non-exhaust particulates from heavier electric vehicles (Beddows and Harrison, 2021) and continue to congest cities with ever increasing traffic volumes. Whilst policy has improved air pollution over recent decades, the UK does not meet World Health Organisation (WHO) guideline values. (WHO Global air quality guideline, 2021) For example, the UK Government recently set a target value for PM_{2.5} of 10 µg m⁻³ to be met by 2040 (The Environmental Targets, 2023) and with exposure to O₃ potentially increasing in cities in future, air pollution outdoors and indoors, will remain a problem in the coming decades.

To investigate the possible co-benefits of NZ policy we have compared future scenarios, assuming existing air quality policy, BAU, with NZ alternatives based upon the CCC's 6th Carbon Budget (The Climate Change Committee). From the 6th Carbon Budget we have used the Balanced Net Zero Pathway (BNZP), a 'middle ambition' pathway for compliance with UK Net Zero commitments by 2050, which is projected to reduce greenhouse gas emissions to 78 % below 1990 levels by 2035, and the Widespread Innovation (WI) pathway, which assumes greater success in reducing costs of low-carbon technologies, allowing more widespread electrification and active transport.

To investigate the co-benefits on air pollution of NZ policy, we predicted PM_{2.5}, NO₂ and O₃ in 2030, 2040 and 2050 using sophisticated air pollution models. Specifically, we compared the air pollution

inequalities, health, and economic impacts of UK BAU with the CCC's BNZP and WI pathways, for road transport, buildings and active travel (i.e. walking and cycling for transport).

2. Materials and methods

2.1. Methodological framework

A project schematic in Fig. 1 describes the process of scenario development including from left to right, emissions development and meteorological model runs (yellow box), through to outdoor and indoor air pollution models (blue box), different physical activity and health scenarios (light green and grey boxes), the assessment of outdoor exposure inequalities (green box) and finally to cost-benefit analysis (purple box). Each aspect of this process is described in more detail in sections 2.2 to 2.5 below.

2.2. Air pollution predictions

We estimated UK wide NO₂, PM_{2.5} and O₃ concentrations in 2019, evaluating our model results against approximately 400 high quality UK measurement sites, from rural to kerbside locations. We included hemispheric contributions to air pollution, used emissions from the Centre on Emission Inventories and Projections in Europe, (CEIP (Centre on Emission Inventories and Projections)) and in the UK, from the National Atmospheric Emissions Inventory (NAEI), (NAEI (National Atmospheric Emissions Inventory), 2019) combined with Imperial's road emissions model. (Beevers et al., 2012) Biogenic emissions were estimated using the Model of Emissions of Gases and Aerosols from Nature (MEGAN v2.1). (Guenther et al., 2012) These were combined with the well-established Weather Research and Forecasting (WRF) meteorological model, (Skamarock et al., 2019) the Community Multiscale Air Quality (CMAQ) model (Byun and Schere, 2006) and Atmospheric Dispersion Modelling System (ADMS) roads model, (CERC, 2021) referred to collectively as CMAQ-urban and allowing predictions every 20 m close to roads. The modelling methods are described in detail elsewhere by Dajnak (Dajnak et al., 2023) with descriptions of the emissions data, air pollution modelling methods and model evaluation used in this analysis, given in Supplementary information (Supp.) S1/2, S3, S4, respectively.

We then predicted NO₂, PM_{2.5} and O₃ concentrations in 2030 and 2040 for three scenarios: BAU, which represented agreed UK air pollution policies and was taken from the NAEI (NAEI (National Atmospheric Emissions Inventory), 2019) and BAU plus two of the CCC's NZ scenarios (BNZP and WI), representing changes to building emissions, through the use of heat pumps, improved energy efficiency and district heat networks, vehicle emissions, through the introduction of electric vehicles and reduced vehicle km travelled, and changes to active travel (cycling and walking). 2030 and 2040 European emissions were scaled from 2019 using emissions trends from the European Clean Air Outlook document (CAO). (European 2nd Clean Air Outlook report) Finally, we predicted pollutant concentrations under a further scenario (BNZP+) in 2050 using BNZP building, vehicles emissions and active travel assumptions in 2050, and for other sources, using European and UK air pollution emissions trends between 2040 and 2050 from the CAO report.

2.3. Inequalities

We assessed inequalities in outdoor air pollution exposure in 2019 and for 2030 and 2040 using the 2011 UK Ward-level Carstairs Index. (Carstairs, 1981) The Carstairs Index combines four dimensions of socioeconomic deprivation: male unemployment, households without car, household overcrowding and head of household in low social class. We quantified NO₂ and PM_{2.5} concentrations by Wards for each scenario, stratified them by Carstairs quintiles and explored inequalities for the UK, each country and region separately.

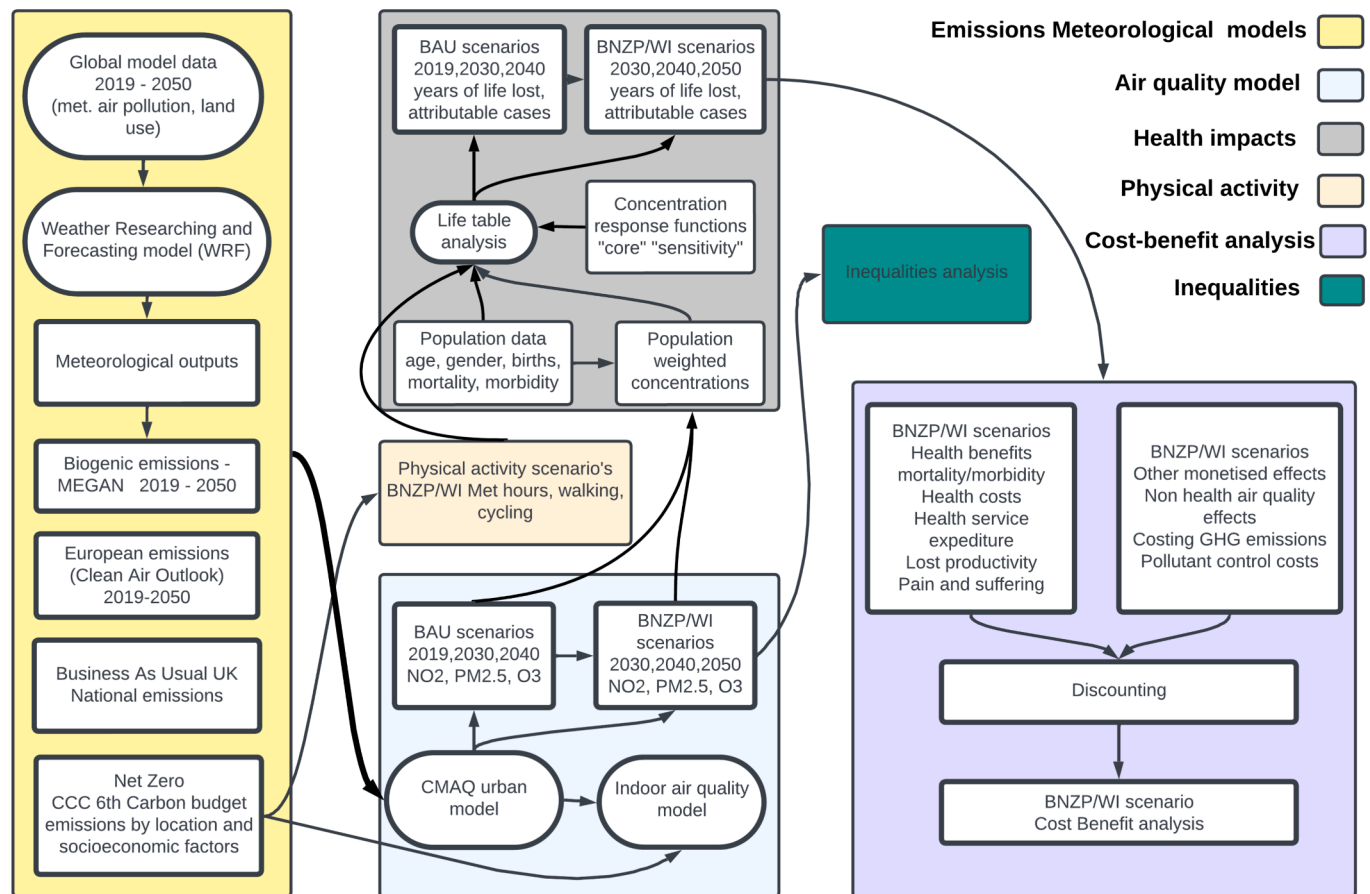


Fig. 1. Project schematic from air pollution emissions and meteorology on the left through air quality models, active travel and health impacts in the middle to inequalities and cost-benefit analysis on the right.

2.4. Health impacts

A complete description of the health impact methods and results using life table analysis, are in an accompanying paper by Walton et al. (Walton et al.) Briefly, to assess the mortality and morbidity outcomes, mapped UK air pollution results from BAU, BNZP, WI and BNZP+ were interpolated to generate annual concentrations covering the period 2019 to 2154, with concentrations after the last modelled year (2040 or 2050) assumed constant through to 2154 (a lifetime after 2050). Recently defined concentration–response functions (CRF) for air pollution; (Forastiere et al., 2024) and physical activity, (Kelly et al., 2014) were combined with population and newly derived baseline rates. Several alternative estimates were made, including a “core” analysis (using better established CRFs), a “sensitivity” analysis, (for the outcomes with reasonable but more uncertain evidence) and with some using ‘cut-off concentrations’ for PM_{2.5} and NO₂, below which we assume no health impact.

The baseline population and health data and birth projections used in our health impact calculations were obtained/derived from multiple sources based on the outcome under investigation, including the Office of National Statistics (ONS), [Statistics.scot.gov](https://statistics.scot.nhs.uk), Northern Ireland Statistics and Research Agency (NISRA), ONS Cancer Registration Statistics, Northern Ireland Cancer Registry, Welsh Cancer Register and Surveillance Unit, Public Health Scotland, the British Lung Foundation, Quality Outcomes Framework and others. More details are provided in Walton et al. (Walton et al.) supplementary material Annex 6.

2.5. Cost benefit analysis

A detailed description of the cost-benefit analysis (CBA) methods is

given in Supp. S5. The CBA was separated into buildings and transport sectors to highlight where air quality makes the greatest economic contribution. Consideration was given separately to the “core” and “sensitivity” health impacts, a summary of which is given in Table S5. Data on capital expenditure (CAPEX, investment in equipment, etc.) and operating expenses (OPEX, or running costs) were taken from the CCC’s 6th Carbon Budget, (The Climate Change Committee) noting that efficiency savings meant that OPEX were negative (i.e. benefits offsetting CAPEX) for both the transport and building sectors. Data on greenhouse gas (GHG) emissions were also taken from the CCC’s 6th Carbon Budget. Methods to account for the health benefits from air pollutant exposure and active travel with costs for loss of utility from disease, are also described in Walton et al. (Walton et al., 2025) Unit healthcare costs, productivity costs, material and ecosystem damage costs and carbon prices (£/t CO_{2e}) are described in Supp. S5. Cost estimates were discounted using UK government guidance (GOV.UK).

Beyond 2050, it was assumed that GHG costs and emissions, and air pollution concentrations remained constant through to 2154, to illustrate the long-term consequences of actions taken up to 2050 but whose impact will persist for many years. It is acknowledged that uncertainty in such long-term projections is large and for that reason the results presented here provide greater resolution in the earlier years. Reasons for uncertainty in the long-term include that benefits of reduced exposure to air pollutants may decline through advances in health science, or costs may decline through improved efficiency and innovation, including existing and new technologies. However, analysis over shorter periods creates its own uncertainty by failing to recognise the long-term consequences of actions taken in the coming decades.

2.6. Indoor air pollution modelling

A detailed description of the indoor modelling methods is given in Supp. S6, but briefly, indoor air pollution was predicted with the CONTAM model, ('CONTAM'. NIST, 13 March, 2012) using outdoor meteorology and air pollution concentrations from WRF and the CMAQ-urban model. London housing characteristics were taken from the UKRI funded WellHome study, (West London Healthy Home and Environment Study) and time spent in each room, indoor sources and use of ventilation, from a survey of 50,000 homes from Imperial's REACT cohort (Imperial College London, 2023). The sampling procedure involved selecting a random sample proportional to the population at the lower-tier local authority level, allowing for representative estimates across England. A total of 14,552 participants responded, resulting in a

response rate of 29 %. CONTAM was developed using methods described by Vu and colleagues (Vu et al., 2022) for a three-bedroom terraced house, representative of 1.5 million homes in London, with no indoor gas cooking sources and evaluated against indoor observations. Gas cooking PM_{2.5} (Sun and Wallace, 2021) and NO₂ (Lebel et al., 2022) emissions were added, and scenarios run with and without this source, with different ventilation strategies and with changes to air tightness associated with improved insulation.

3. Results

3.1. Outdoor air pollution concentrations

The switch to cleaner vehicles between 2019 and 2030 under BAU

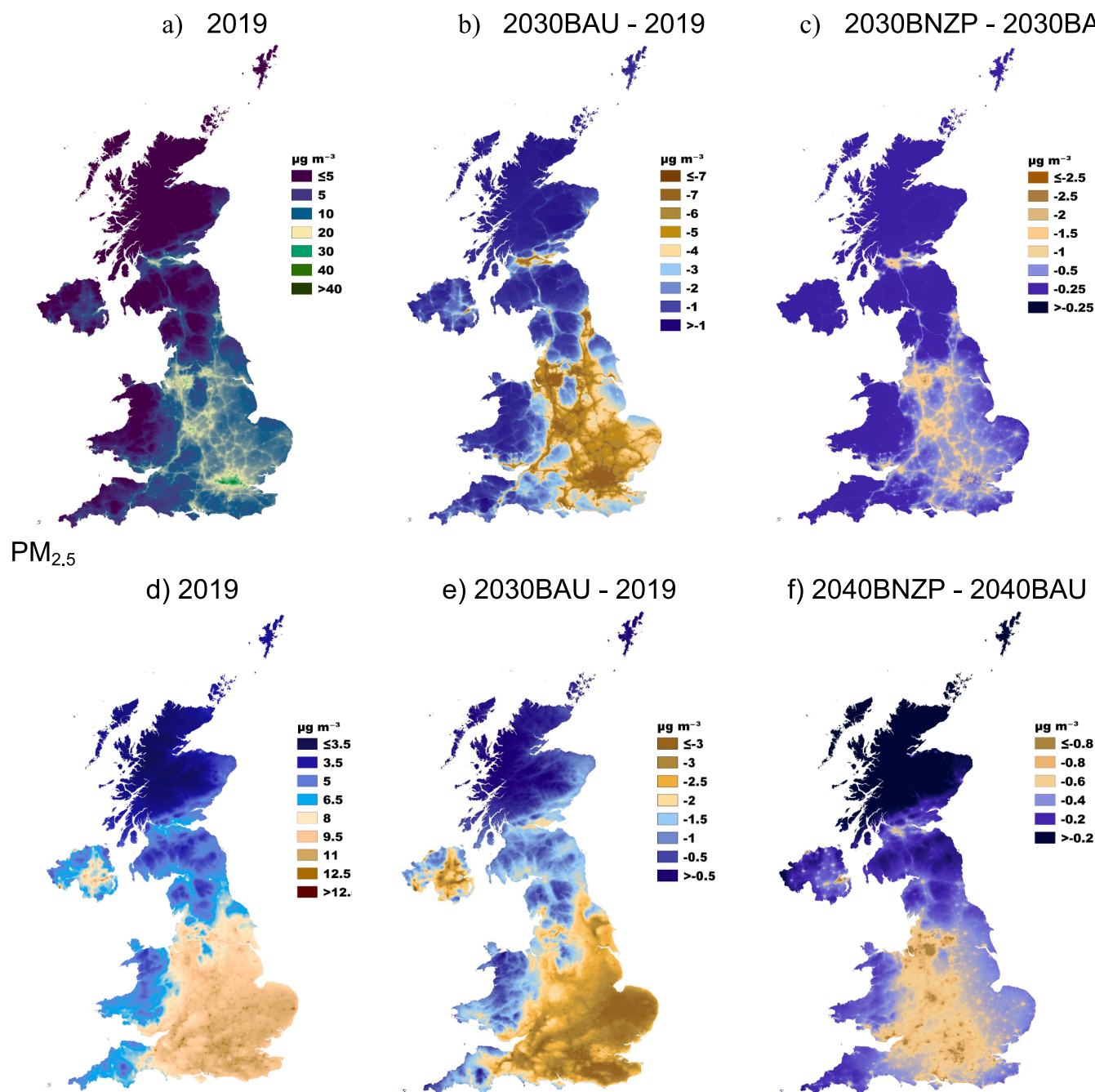


Fig. 2. The 2019 annual average concentrations of NO₂ (top) and PM_{2.5} (bottom) (µg m⁻³) and difference in concentrations between 2030 and 2040 BAU and BNZP scenarios.

(Fig. 2b) and the further integration of electric vehicles (EV) and low carbon building heating under BNZP (Fig. 2c), reduced average UK concentrations of NO_2 , with the largest changes in urban areas and close to major roads. When weighted by population the average UK concentrations reduced by $6 \mu\text{g m}^{-3}$ (2030 BAU vs. 2019) and by a further $0.8 \mu\text{g m}^{-3}$ (2030 BNZP vs. 2030 BAU). Model source apportionment (Fig S6) showed reductions in domestic heating NO_x emissions in the BNZP becoming increasingly important by 2040. Under BAU assumptions, between 2019 and 2030 (Fig. 2e) $\text{PM}_{2.5}$ concentrations reduced significantly, but changed very little between 2030 and 2040 (not shown). However, by 2040, (Fig. 2f) under BNZP, $\text{PM}_{2.5}$ reduced, once again with the largest changes in urban areas. When weighted by population the average UK concentrations reduced by $3 \mu\text{g m}^{-3}$ (2030 BAU vs. 2019) and by a further $0.7 \mu\text{g m}^{-3}$ (2040 BNZP vs. 2040 BAU). Source apportionment (Fig S6) showed EV uptake to represent approximately 24 % and building heating the remaining 76 % of the $\text{PM}_{2.5}$ change. The UK concentrations, in Fig. 2, were used as part of the inequalities, health impacts, CBA and indoor modelling, given in sections 3.2 to 3.5, below.

3.2. Inequalities

We observed on average $\sim 5 \mu\text{g m}^{-3}$ higher NO_2 concentrations in the most deprived (quintile 5) compared to the least deprived (quintile 1) Wards in 2019 across the UK (Fig. 3). Inequalities in NO_2 concentrations lessened with decreasing air pollution levels in 2030 and 2040 for all scenarios, but most deprived areas still had the highest air pollution concentrations. For $\text{PM}_{2.5}$, the pattern across the socioeconomic spectrum was less clear with similar concentrations across the Carstairs quintiles. The linear trends in inequalities observed for NO_2 in the UK were similar to those seen for England, while trends in the devolved nations followed a J-shape or U-shape distribution with lowest concentrations observed in the mid-deprivation strata (see Supp. S7). Air pollution inequalities were greatest in Scotland with differences between least and most deprived areas up to $\sim 7 \mu\text{g m}^{-3}$ for NO_2 (see Supp. S7).

3.3. Health impacts and active travel results used in the CBA analysis

Given the potential for double counting between NO_2 and $\text{PM}_{2.5}$, our health impact assessment for mortality focused on the effects of $\text{PM}_{2.5}$,

which gave the largest impact in 2019. The difference between BNZP $\text{PM}_{2.5}$ concentrations compared with BAU resulted in 3.8 (95 % CI 2.9 to 4.3) million life years gained (LYG) over the period 2019–2154. Changes in active travel by walking or cycling for BNZP vs BAU added a further 1.1 (0.7 to 1.6) million LYG giving an overall total of 4.9 (4.1 to 5.8) million. For the WI scenario vs BAU, greater changes in physical activity resulted in 1.7 (1.0 to 2.5) million LYG, while LYG for $\text{PM}_{2.5}$ reductions were similar to BNZP vs BAU (3.7 (2.8 to 4.1) million). We also assessed morbidity, including multiple health outcomes, which resulted in large health impacts and monetised benefits. For BNZP vs BAU, the air quality benefits up to 2154 were £52.1 billion (39.5 to 64.7) for morbidity and £77.9 billion (42.9 to 90.8) for mortality, while the corresponding numbers for WI were £50.0 billion (37.9 to 62.1) and £74.7 billion (39.4 to 84.4), respectively. Since mortality and most morbidity benefits were driven by $\text{PM}_{2.5}$ concentration changes, in the BNZP scenario these were mostly driven by building emissions. A full description of the Section 3.3 numbers is in the accompanying paper Walton et al. (Walton et al.).

3.4. CBA for the BNZP pathways

Fig. 4 shows how net economic benefits accrue over time for the buildings (top panel) and transport (bottom panel) sectors for the BNZP scenario, as benefits are successively added to offset CAPEX. Analysis shown in the figure starts from 2019 and runs to 2060. Benefits will continue to accrue beyond that point, though with greater uncertainty. Recognising this, results extended to 2154 (covering the lifespan of the population affected by policies up to 2050) are presented in (Supp. S8). Operating costs (OPEX) show as benefits because of reduced running costs via efficiency improvements for both sectors. This is particularly apparent for transport, where reduced running costs alone generate a net benefit after only 20.5 years (see also Table 1). For the buildings sector, OPEX was not large enough on its own to exceed CAPEX and generate an overall net benefit. Adding the benefits of reduced greenhouse gas (GHG) emissions next (given that these policies were directed at climate mitigation) was sufficient to generate an overall net benefit for the domestic sector after between 17.8 to 33.5 years (Table 1) depending on the valuation of GHG reduction benefits. Improved air quality and increased active travel provided additional justification for the CCC scenarios and was most evident for the buildings sector when applying lower bound estimates of GHG reduction benefits. In this case,

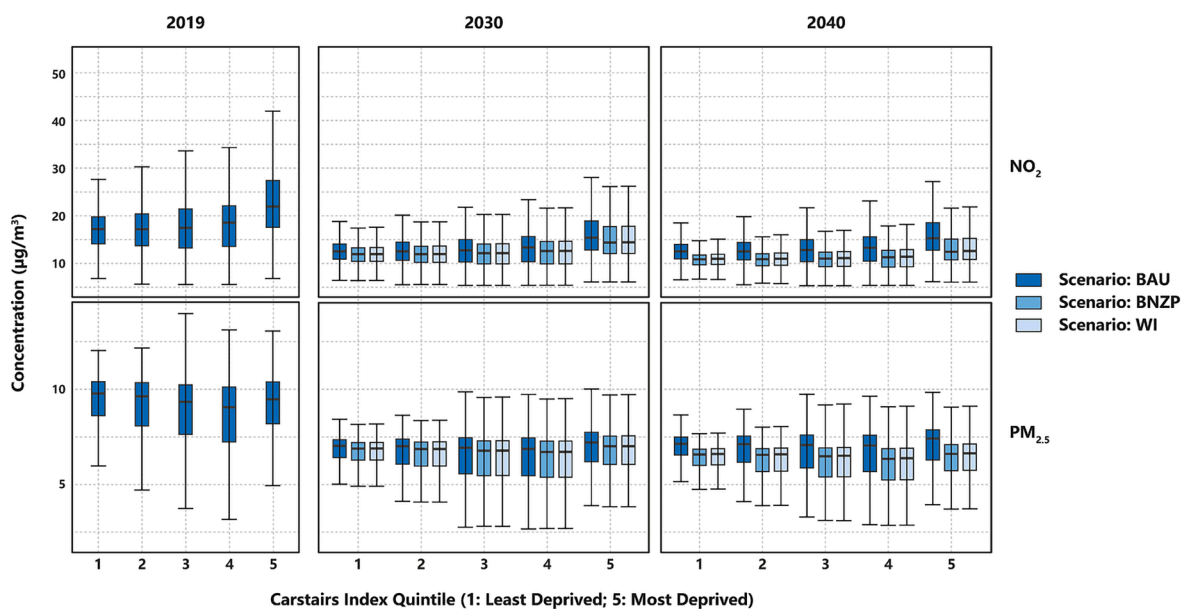


Fig. 3. Ward-level Nitrogen dioxide (NO_2) and fine particulate matter ($\text{PM}_{2.5}$) concentrations by Carstairs quintiles (Quintile 1: least deprived, quintile 5: most deprived) across the UK in 2019 and predicted concentrations based on BAU, BNZP and WI pathways for 2030 and 2040.

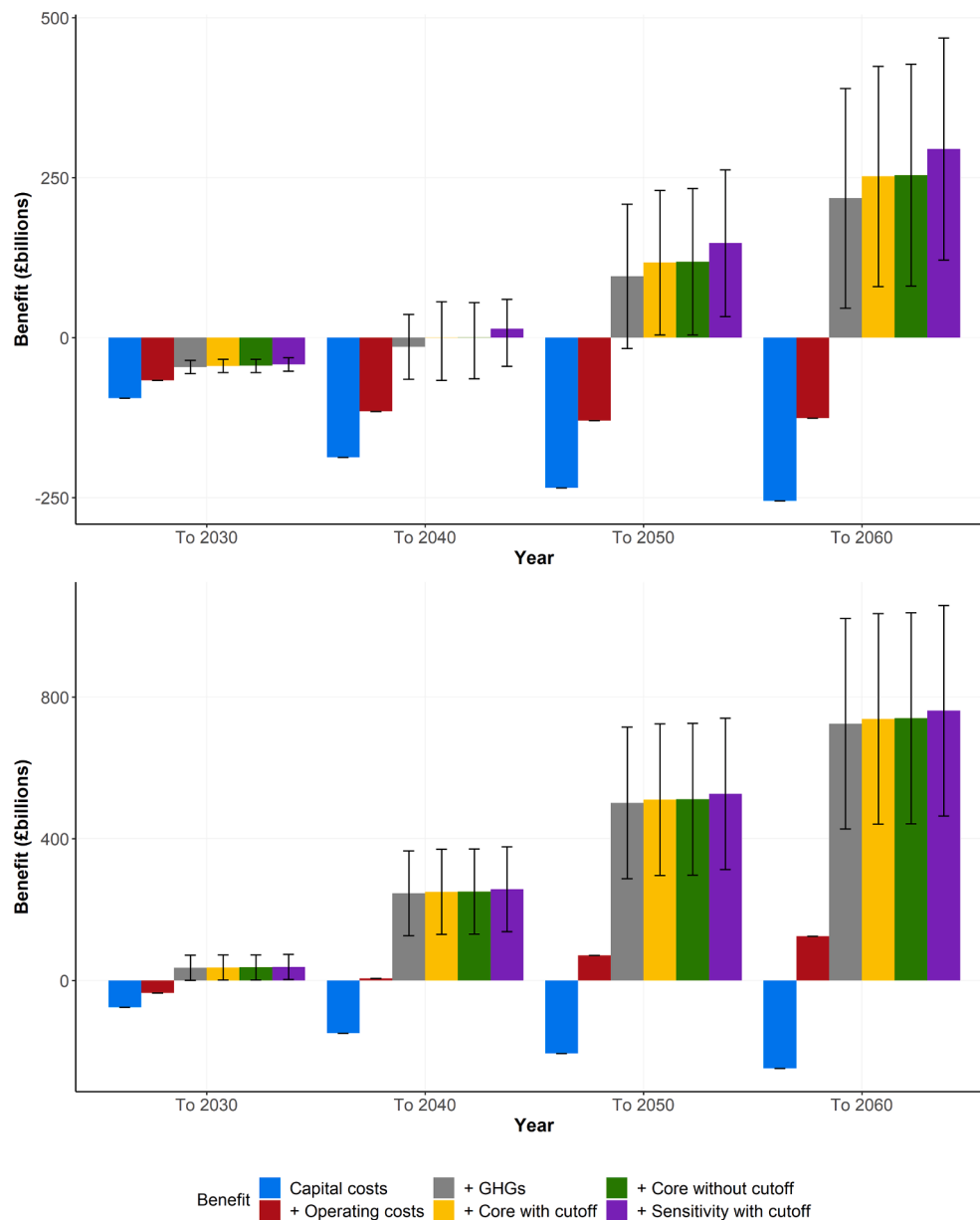


Fig. 4. Net economic benefits for the buildings sector under the BNZP scenario (top panel) and transport sector under the BNZP scenario (bottom panel) as benefits are successively added to capital costs.

air pollution benefits were sufficient to advance time to break-even by 3.1 (2.5 to 4.7) years when including “core” health effects, or 6.3 (4.7 to 7.6) years when adding in the “sensitivity” effects, or from 2052 to between 2046 and 2049. Assuming larger GHG benefits, the influence of air pollution and active travel on time to break-even declines but is still present.

Reflecting the results in Fig. 4 (see also Supp. S8), the benefit-cost ratio (BCR) for the buildings sector is dominated by GHG mitigation benefits. Stripping these benefits out shows that by 2050, core health effects add 0.09 (0.07 to 0.12) to the BCR, indicating that core air pollution benefits offset 9 % of costs, increasing to 0.21 (0.15 to 0.28) when sensitivity effects are included. Adding in reduced OPEX via efficiency savings (but still not accounting for GHG mitigation benefits) increases the BCRs to 0.54 (0.51 to 0.57) and 0.66 (0.59 to 0.74), respectively. Over longer periods the air quality health benefits become more significant, increasing by 2154 to 0.31 (0.25 to 0.37) for core effects only and 0.63 (0.46 to 0.80) when adding sensitivity effects (Supp. S8).

In contrast, the effects of accounting for air pollution and active travel benefits on time to break-even for the transport-BNZP scenario are more modest, given that operating costs and GHG reduction benefits lead to a net benefit after a much shorter time, of 10.9 years under the lower bound GHG benefit case. This falls to 7.1 and 2.0 years when adopting the central and high values for GHG benefits respectively, a sufficiently short period that the lags associated with long term air pollution health impacts mean that health benefits of the transport policies were still to be fully realised, with some effects taking up to 20 years. Core and sensitivity effects added 0.06 (0.05 to 0.08) and 0.14 (0.07 to 0.18) to BCR by 2050 and 0.16 (0.14 to 0.18) to 0.30 (0.14 to 0.35), respectively by 2154. Results for the transport-WI scenario were broadly similar (Supp. S8), with added benefits of more active travel partially offset by a small decrease in air pollution health benefits relative to the BNZP scenario. Overall BCR for the WI scenario is higher, but as a consequence of reduced capital costs linked to innovation assumptions. A key uncertainty for the transport sector concerns assumptions regarding the treatment of mortality for which preference has

Table 1

Time (years) for benefits to match costs (break-even point) for the buildings and transport sectors under the BNZP scenario. ‘Lower/Upper health estimates’ factor in the lower and upper confidence intervals, respectively from health impact analysis.

Years to break-even	Lower health estimate	Central health estimate	Upper health estimate
Buildings sector			
+ operating costs	no break even	no break even	no break even
+ GHG lower benefits	33.5	33.5	33.5
+ air pollution health benefits			
+ core with cut-off	31.0	30.4	30.0
+ sensitivity with cut-off	28.8	27.2	25.9
+ GHG central benefits	22.8	22.8	22.8
+ air pollution health benefits			
+ core with cut-off	21.8	21.6	21.4
+ sensitivity with cut-off	20.9	20.3	19.7
+ GHG upper benefits	17.8	17.8	17.8
+ air pollution health benefits			
+ core with cut-off	17.3	17.2	17.1
+ sensitivity with cut-off	16.8	16.4	16.1
Transport Sector			
+ operating costs	20.5	20.5	20.5
+ GHG lower benefits	10.9	10.9	10.9
+ air pollution and active travel health benefits			
+ core with cut-off	10.8	10.8	10.8
+ sensitivity with cut-off	10.7	10.6	10.6
+ GHG central benefits	7.1	7.1	7.1
+ air pollution and active travel health benefits			
+ core with cut-off	7.0	7.0	6.9
+ sensitivity with cut-off	6.9	6.8	6.8
+ GHG upper benefits	2.0	2.0	2.0
+ air pollution and active travel health benefits			
+ core with cut-off	1.9	1.9	1.9
+ sensitivity with cut-off	1.9	1.9	1.9

been given to quantification against PM_{2.5} for reasons discussed by Walton et al. (Walton et al.) For the transport sector, the mortality related benefits of reductions in NO₂ would be twice those for PM_{2.5} (see Supp. S8) given the larger changes in NO₂ relative to PM_{2.5}. This would increase benefits but have little impact on the conclusions of the CBA

because of the dominance of changes in GHG emissions. The same does not apply to the buildings sector where changes in NO₂ are much less than for PM_{2.5}.

Information on other CBA sensitivities such as the effect of applying cut-off concentrations to the analysis and exclusion of response functions for some pollutants due to concerns over potential double counting, is provided in Supp. S8. Both sensitivities were found to have the potential to affect conclusions in the assessment of air pollution policies, though in the context of climate policy analysis and given the importance of other benefits such as efficiency savings and GHG reductions, these effects were less prominent.

BNZP+ scenario: This scenario includes additional controls for all sources, targeted at regional air pollutants from the EU as well as the UK and extending the time frame for introduction of those measures from 2040 to 2050. This highlights wider potential for reducing air pollution in the UK than represented by the CCC BNZP and WI scenarios, alone. The additional air pollution health benefits of the BNZP+ scenario for all sectors combined compared to BNZP alone are shown in Fig. 5. There is no difference in the early years when the same assumptions are used for both scenarios. Thereafter, additional benefits of BNZP+ grow rapidly, highlighting the potential for reducing air pollutant emissions beyond that captured by the CCC scenarios up to 2040. BNZP+ air quality benefits relative to BAU accrued to 2060 were 2.2 times higher (for both core effects and core + sensitivity effects) than for BNZP (by 2154, accepting increased uncertainty, these figures rise to 3.1 and 3.2 times higher, respectively). A comparison of costs and benefits for this scenario is not possible given a lack of cost data for some of the emission reductions such as those from the EU Member States. The benefits of the scenario at a European scale are underestimated because analysis is focused on health impacts within the UK, not accounting for effects in other countries.

3.5. NZ impacts in the home

The CCC 2040 BNZP scenario in UK homes requires large scale improvements in insulation, increasing air tightness, electrification of heating/removal of fossil fuels and district heating. Potential air pollution exposure impacts were estimated by removing gas cooking and improving home insulation (see Fig. 6), whilst varying levels of home ventilation for a representative London household. The left-hand side of Fig. 6 represents increased indoor air pollution in a home with gas cooking that has insulation installed up to at least an Energy Performance Certificate (EPC) rating of C, the minimum standard required for

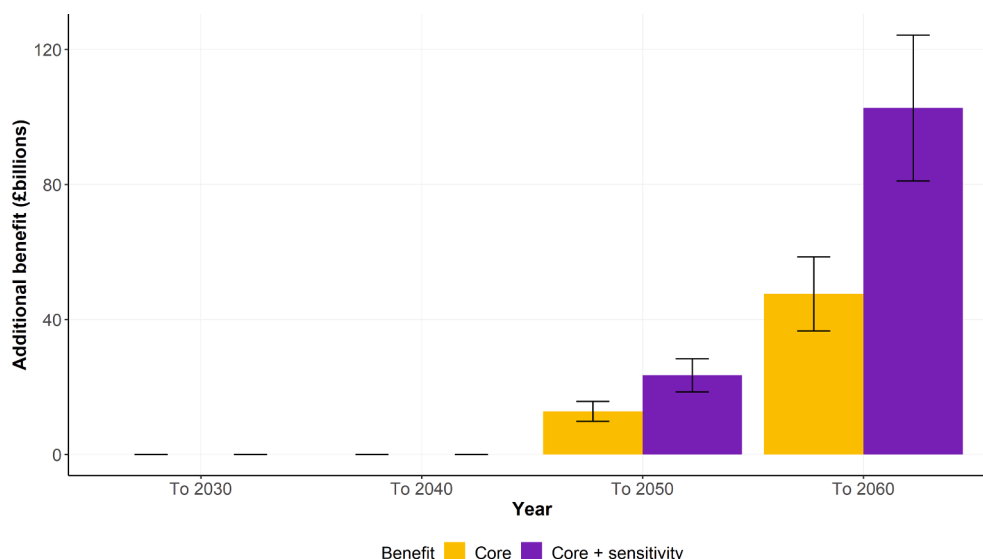


Fig. 5. Additional air pollution health benefits for “core” effects only and for “core” + “sensitivity” effects of the BNZP+ scenario relative to BNZP.

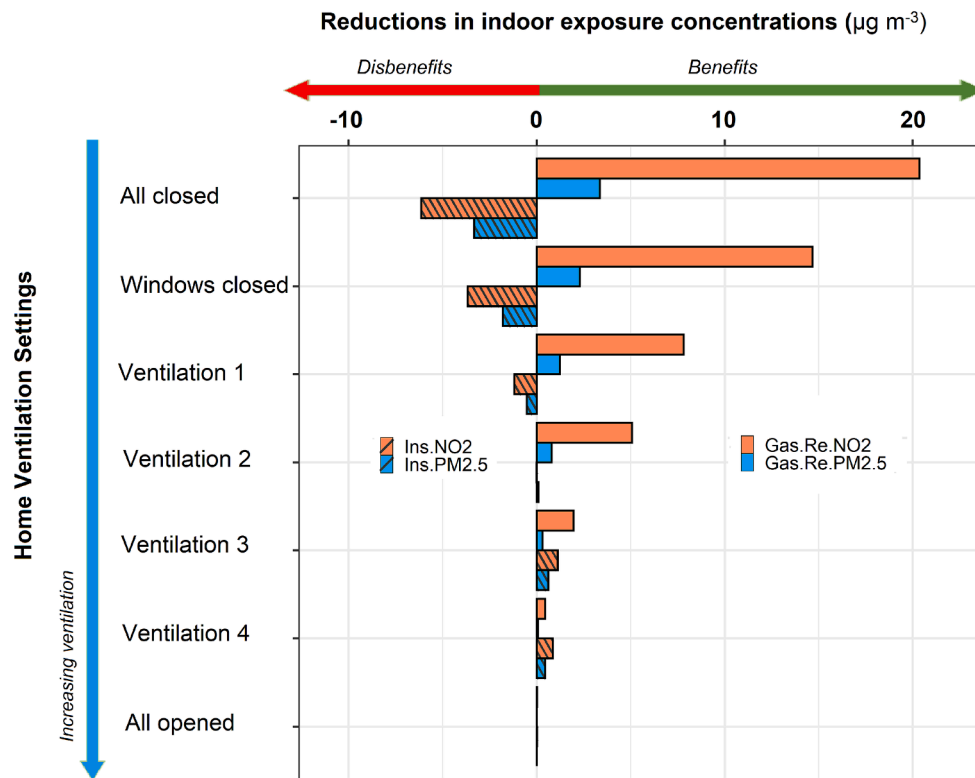


Fig. 6. Changes in indoor exposure concentrations of PM_{2.5} and NO₂ due to home insulation (Ins.) and gas cooking removal (Gas.Re.) with various home ventilation settings. Ventilation scenarios: *All closed*: Cooking hoods and windows were closed; *Windows closed*: all windows were closed, except cooking extractor fans; *Ventilation 1*: Rear kitchen window was opened during cooking; *Ventilation 2*: Rear kitchen and front lounge windows were opened during cooking; *Ventilation 3*: Windows in kitchen, living, and bedroom were opened during cooking; *Ventilation 4*: Windows in kitchen were opened all the time while others are opened during cooking. *All opened*: All windows were opened all the time.

NZ, and the right-hand side, after installing insulation, shows the effects of removing gas cooking. The ventilation settings ranged from an 'All closed' scenario, with very limited ventilation to an 'All opened' scenario, with homes permanently ventilated and four intermediate strategies. Upgrading insulation to EPC C or above, from EPC D/E, applies to 9.5 million UK homes. We assumed 2040 outdoor air pollution concentrations of PM_{2.5} of 7.2 µg m⁻³ and NO₂ of 12.1 µg m⁻³.

Upgrading home insulation to meet NZ requirements benefited homes without gas cooking, due to protection from outdoor air pollution ingress, and reduced indoor exposure to PM_{2.5} in 2040 by 1.3 µg m⁻³ and NO₂ by 2 µg m⁻³. Where homes have gas cooking, the increased air tightness comes with disbenefit for indoor environments from the 'All closed' scenario resulting in a 3.3 µg m⁻³ PM_{2.5} and a 6.1 µg m⁻³ NO₂ increase, the 'Windows closed' scenario resulting in a 1.8 µg m⁻³ PM_{2.5} and 3.7 µg m⁻³ NO₂ increase and the 'Ventilation 1' scenario resulting in a 0.5 µg m⁻³ PM_{2.5} and 1.5 µg m⁻³ NO₂ increase in exposure. Increasing ventilation further removes the disbenefit of improving insulation by ventilating indoor pollutants generated from gas cooking outdoors and despite enhancing the ingress of outdoor air pollution, there is a small benefit for both ventilation scenario 3 and 4."

Assuming the household is insulated to NZ standard, removing gas cooking has the effect of reducing exposure for all ventilation settings, with the largest effects in the 'All closed' scenario, which resulted in PM_{2.5} reductions of 3.4 µg m⁻³ and for NO₂ of 20.4 µg m⁻³, whilst for the Windows closed scenario, PM_{2.5} reduced by 2.3 µg m⁻³ and NO₂ reduced by 14.7 µg m⁻³. For the Ventilation 1–4 scenarios, a reduction of indoor exposure ranged from 0.1 – 1.2 µg m⁻³ for PM_{2.5} and 0.4 – 7.8 µg m⁻³ for NO₂.

4. Discussion

This study contributes to the evidence base of air pollution and health costs and benefits of meeting UK Net Zero targets related to road transport, building heating and active travel. It includes both mortality and a comprehensive set of morbidity health benefits linked to reductions in air pollution, mortality benefits of active travel, CAPEX and OPEX costs and GHG impacts. It also addresses current and future air pollution inequalities and the importance of the home environment for human exposure. The novelty of this work includes: Its use of highly complex air pollution models for future predictions, with contributions (emissions and meteorology) at hemispheric, European and UK scale; comparison of four realistic future cases, a UK Business as Usual case, which includes currently agreed environmental policy with published NZ scenarios from the CCC's 6th Carbon Budget; quantification of changes in disease incidence effects, which are rarely assessed; detailed economic appraisal; and changes in air quality inside housing, where NZ changes to air pollution exposure may be larger than those outdoors. As such the results and methods presented here are applicable to UK policymakers and to many developed countries facing the transition of transport and building sectors to achieve NZ targets.

An extensive review by Karlsson et al., (Karlsson et al., 2020) highlights that research has repeatedly established the importance of air quality co-benefits to policy appraisal, although this broadly agrees with the conclusions of this study, a direct comparison is difficult due to the wide and differing range of assumptions and scenarios. Our study also agrees with earlier work by Markandya et al., (Markandya et al., 2018) who demonstrated that the mortality co-benefits from air pollution reductions linked to climate mitigation measures are both large and economically important. Once again however, a direct comparison between Markandya's results and ours is difficult, as the former study

estimated co-benefits in India, EU-27, USA, China and the rest of the world, but not specifically in the UK.

Our analysis differs in detail from Milner et al., (Milner et al., 2023) for transport, indoor environments, and active travel, by considering a broader range of health impacts of air pollution, a longer timescale (2154 vs. 2100), the whole UK (rather than only England and Wales) and using BAU exposures beyond 2020 (the point of comparison used by Milner). In addition, we did not undertake a health impacts assessment indoors, unlike Milner, due to the large uncertainties involved in doing so. This is despite demonstrating the potentially large reductions in air pollution exposure, which would have increased the health benefits further. Because of these methodological differences, it is difficult to compare results directly. However, the two studies agree on the broad scale of co-benefits from the climate policies considered, and both conclude that they are substantial, and hence are important to include in future economic policies of possible climate actions.

Our finding, that UK NZ policies have larger mortality benefits from reduced exposure to PM_{2.5} than increased physical activity may seem at odds with other work on transport health impacts, which typically find the physical activity benefits to be many times larger than the air pollution impacts. (Mueller et al., 2015 Jul) This reflects the fact that the CCC NZ scenarios modelled a wide range of air pollution reduction policies (including electrification of the vehicle fleet, rather than just mode shift from cars to active travel), while also being relatively unambitious in terms of mode shift to active travel, especially for walking for transport. (Aldred et al., 2024).

Our results generally agreed with Wilkinson et al (Wilkinson et al., 2009), on the health benefits of home ventilation and fuel switching, although Wilkinson used a hypothetical housing scenario and calculated health benefits related to PM_{2.5}, Radon, CO, second hand tobacco smoke, Mould and cold, all of which make it difficult to make a meaningful direct comparison.

Williams et al., (Williams et al., 2018) estimated 1.1 and 0.5 million LYL from PM_{2.5} between 2011 and 2154 in Great Britain (for the nuclear power and low-greenhouse gas scenario respectively, compared with the BAU), and that this was caused by a peak of biomass (wood burning) in 2035 and by a large projected increase in future demand for transport, leading to potential increases in non-exhaust particulate matter emissions. However, such large-scale biomass burning is no longer considered to play a significant role in NZ, mainly featuring in power generation, and with domestic use superseded by heat pumps, district heat networks and use of limited amounts of hydrogen. In addition, whilst concerns remain around non-exhaust transport emissions, future NZ scenarios are now associated with a reduction in vehicle km, and that by using the novel work of Hicks, (Hicks et al., February 2021; Hicks et al., 2023) on the benefits of regenerative braking, we estimated that light-duty EVs are beneficial for non-exhaust PM_{2.5}, despite their increased weight.

Since the health impacts of NZ policy were primarily driven by PM_{2.5}, we used a state-of-the-science chemical transport model (CTM), CMAQ-urban, described elsewhere by Dajnak et al, (Dajnak et al., 2023) and supported by model evaluation in Supp. S4. PM_{2.5} is a spatially and temporally complex mixture that travels long distances across the UK and Europe. It combines emissions from both primary sources (e.g. vehicle exhaust) and secondary sources of gaseous pollutants that form PM through atmospheric chemistry and is both anthropogenic and biogenic in origin. Exposure was via predictions every 20 m close to roads to represent highly spatially varying pollutants such as annual average NO₂ and hourly O₃ predictions, required for the health impacts. Whilst uncertainty in the modelling is acknowledged, results provide a current state-of-the art view on how concentrations and exposures will change in the coming years.

Detailed emissions modelling allowed us to implement NZ geographically and with consideration of socioeconomic factors. For example, installing low carbon heating prioritised homes with poor energy efficiency, the fuel poor, those least able to pay and in the private

rented sector, working towards the wealthiest owner-occupiers. This better reflected an equitable policy whereby those most in need are helped earlier to make the transition towards NZ. This may explain some of the reduction in air pollution inequalities which we observed under NZ scenarios, and by dedicating policies to specific socioeconomic groups could substantially reduce other health inequality gaps such as cold, damp and mouldy homes.

The changes in indoor air pollution exposure due to NZ policies were large in certain cases and demonstrate what is possible for low carbon homes. However, for a complete assessment of changes to indoor exposure, a UK wide model would need to be established and able to account for different house types, geographic locations, indoor environmental factors (such as heating and ventilation systems) and behaviour related to ventilation settings, cooking, smoking and other indoor activities. Such large reductions should encourage a UK modelling effort in future research.

Health impact sensitivity demonstrates that we have taken a conservative approach, with potential for significant additional benefits. For example, impacts on dementia and diabetes were included in sensitivity rather than core analysis. Furthermore, mortality was only for exposure to PM_{2.5}, although evidence for additional pollutant effects, particularly NO₂, exists. The extent to which PM_{2.5} CRFs account for NO₂ effects and vice-versa is unknown. Further analysis presented in Supp. S8 indicates that transport benefits particularly, would increase if greater weight were given to effects quantified against NO₂ exposure.

Extension of analysis over a long time frame (here, to 2154) is not unusual in health impact assessment. (Committee on the Medical Effects of Air Pollutants (COMEAP), 2018) However, CBA is typically carried out for shorter periods (e.g. to 2050). Here, we provide CBA results over different timescales, showing how the economic case for change develops over time. This analysis highlights that the fundamental changes made to transport and domestic energy use under NZ will have long lasting benefits for the UK. Increased uncertainty over longer time periods is acknowledged, and this may take several forms, for example, on the one hand the economic burden of future health impacts may decline as new treatment options become available, but on the other, future technologies that are more efficient may be expected to bring down costs. Whilst uncertainties are naturally present, they are insufficient to undermine the conclusion that the health and economic benefits to health linked to air pollutant reductions and increases in active travel from climate policies are significant and should be accounted for in development of those policies. Failure to account for them could limit the extent to which potential improvements are exploited, weakening the case for climate action.

Our analysis indicates considerable additional benefits from removing indoor sources. Whilst we acknowledge that available response functions for outdoor air from the epidemiological literature will include some effects of exposure to outdoor sources indoors, there are considerable uncertainties in separately quantifying the health impacts of indoor air pollution, particularly for PM_{2.5}. (Committee on the Medical Effects of Air Pollutants (COMEAP), 2023) Consequently, we were unable to quantify or monetise these benefits, pointing to the need for future research in this area. Inclusion of additional benefits from reductions in indoor pollution would have further strengthened the economic case for controls on fossil fuel use in buildings.

In summary, this study agrees with others that the co-benefits of air pollution and health associated with NZ are large and important and that they contribute to the economic justification of NZ, especially in the buildings sector. Whilst we have used state of the science models to make future outdoor air pollution predictions and have rightly applied NZ changes in homes based upon socioeconomic factors, we have also pointed to the need for a comprehensive assessment of UK indoor air pollution and its health effects, as it could bring significant additional benefits. We have undertaken health impacts modelling under a range of assumptions, with our economic assessment demonstrating the importance of considering a wide range of morbidity impacts, even at what

are, historically, low concentrations. This has pointed to the need for further health evidence for outcomes such as dementia and diabetes both of which were given a reduced confidence rating by Forastiere et al. (Forastiere et al., 2024) Whilst most health impact assessments focus on PM_{2.5}, the importance of NO₂ should not be underestimated given the certain benefits of reduced emissions from vehicle electrification, and from controls within the home. As such our analysis may underestimate the NO₂ effect. Also, that research of this kind should be policy relevant, meaning comparing published NZ projections with existing policies, such as BAU, and avoiding hypothetical scenarios where possible. However, we should also be mindful that the success of existing BAU policies is far from guaranteed, and that by eliminating fossil fuels and increasing active travel, NZ provides a potentially important alternative. Finally, outdoor air pollution exposure inequalities were predicted to diminish in future and that NZ contributed to this change. However, indoor air pollution exposure inequalities remain an area of concern that is rarely if ever considered in studies of this kind.

5. Conclusions

What is striking from the NZ transition, especially for buildings, is the scale of change required in the coming decades, with millions of homes requiring new insulation and heat pumps. The House of Lords Environment and Climate Change Committee have commented that NZ targets are unlikely to be met (The Lords Environment and Climate Committee, 2023). This points to the need for considerable public engagement, to increase awareness of associated issues and benefits, to encourage political ambition on environmental policies, technology, and incentives, and to create a convincing narrative for, and commitment to, the large scale of change that is needed.

Our results forecast important economic benefits of removing fossil fuel burning whilst improving energy efficiency in UK homes. Whilst UK targets for both insulation and heat pump installations exist, this research should encourage more ambitious policy, especially targeted to help those most in need in society and to address affordability.

The CBA showed that NZ changes to the vehicle fleets were cost effective and that air pollution health impacts improved things further. Our core analysis provides a possibly conservative view on air pollution benefits, with a preference for estimates linked to NO₂ rather than PM_{2.5}, which would have seen them doubled. This should encourage ambitious policies to ensure the EV transition continues.

We found that meeting road transport net zero relies on increasing active travel to reduce future travel demand and improve city congestion. Yet whilst the CCC's scenario projected a significant increase for future cycling, scenario levels of walking barely changed and are deserving of more attention.

Recognition of co-benefits is an important element in achieving efficient use of resources across policy areas, with this study highlighting this across climate, air pollution, exercise and inequalities.

6. Data sharing

The base year 2019 emissions (<https://naei.beis.gov.uk/>) and UK air pollution measurements (<https://uk-air.defra.gov.uk/networks/network-info?view=aurn>) are freely available, as are the WRF (<https://www.mmm.ucar.edu/models/wrf>) and CMAQ model code (<https://www.epa.gov/cmaq/access-cmaq-source-code>). Future changes in emissions are provided in the [supplementary information](#). Maps of results by local authority are also available upon request. The Small Area Health Statistics Unit does not have permission to supply data to third parties.

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CRediT authorship contribution statement

Sean Beevers: Writing – original draft, Validation, Supervision, Project administration, Methodology, Investigation, Funding acquisition, Conceptualization. **Nosha Assareh:** Writing – review & editing, Visualization, Validation, Software, Methodology, Investigation, Formal analysis, Data curation. **Andrew Beddows:** Writing – review & editing, Validation, Methodology, Investigation, Formal analysis, Data curation. **Gregor Stewart:** Writing – review & editing, Validation, Investigation, Formal analysis, Data curation. **Mike Holland:** Writing – original draft, Validation, Methodology, Investigation, Formal analysis, Data curation, Conceptualization. **Daniela Fecht:** Writing – original draft, Validation, Investigation, Formal analysis, Data curation, Conceptualization. **Yunzhe Liu:** Writing – review & editing, Validation, Methodology, Investigation, Formal analysis, Data curation. **Anna Goodman:** Writing – review & editing, Validation, Investigation, Data curation. **Heather Walton:** Writing – original draft, Supervision, Methodology, Investigation, Formal analysis, Data curation, Conceptualization. **Christian Brand:** Writing – original draft, Validation, Methodology, Investigation, Formal analysis, Data curation, Conceptualization. **Dimitris Evangelopoulos:** Writing – review & editing, Validation, Methodology, Investigation, Formal analysis, Data curation. **Dylan Wood:** Writing – review & editing, Validation, Investigation, Formal analysis, Data curation. **Tuan Vu:** Writing – original draft, Validation, Methodology, Investigation, Formal analysis, Data curation. **David Dajnak:** Writing – review & editing, Validation, Methodology, Investigation, Formal analysis, Data curation.

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.

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Appendix A. Supplementary data

Supplementary data to this article can be found online at <https://doi.org/10.1016/j.envint.2024.109164>.

Data availability

Data will be made available on request.

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