Who benefits from environmental policy? An environmental justice analysis of air quality change in Britain, 2001–2011

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**LETTER**

# Who benefits from environmental policy? An environmental justice analysis of air quality change in Britain, 2001–2011

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## Abstract

Air quality in Great Britain has improved in recent years, but not enough to prevent the European Commission (EC) taking legal action for non-compliance with limit values. Air quality is a national public health concern, with disease burden associated with current air quality estimated at 29 000 premature deaths per year due to fine particulates, with a further burden due to $NO_2$. National small-area analyses showed that in 2001 poor air quality was much more prevalent in socio-economically deprived areas. We extend this social distribution of air quality analysis to consider how the distribution changed over the following decade (2001–2011), a period when significant efforts to meet EC air quality directive limits have been made, and air quality has improved. We find air quality improvement is greatest in the least deprived areas, whilst the most deprived areas bear a disproportionate and rising share of declining air quality including non-compliance with air quality standards. We discuss the implications for health inequalities, progress towards environmental justice, and compatibility of social justice and environmental sustainability objectives.

## 1. Introduction and background

### 1.1. Air quality and disease burden in Great Britain (GB)

Outdoor air pollution makes a significant contribution to mortality in the UK, greater than that from either second hand smoking or road traffic accidents (COMEAP 2010). Globally, it ranks ninth out of 67 health risk factors (Lim *et al* 2012). First estimates of the UK burden of disease, as premature deaths, made by the Committee of Medical Experts on Air Pollution (COMEAP), were 8 100 deaths due to $PM_{10}$ (fine particulate matter $<10\ \mu m$ diameter), 3 500 deaths due to sulphur dioxide, and 700–12 500 deaths for ground level ozone (COMEAP 1998). However COMEAP recognized these as under-estimates as they related only to short term episodic exposure, omitting chronic (long term, low level) exposure. More recently, improved epidemiological evidence has permitted an estimate of the overall UK burden of disease attributable to long term exposure to particulate concentrations (as $PM_{2.5}$). The central estimate is 29 000 premature deaths in 2008, with a range of 4 700–51 000 (COMEAP 2010). This estimate received considerable media attention, yet COMEAP explain how the estimates are based on statistical life years, with the 29 000 figure reflecting a situation where these were the only people affected, each losing 11.5 years of life. This is unlikely in practice, so COMEAP provide alternative interpretations of the disease burden, including 200 000 people dying two years prematurely in 2008, or everyone born in 2008 having a lifespan reduced by six months.

COMEAP's disease burden estimate ignores nitrogen dioxide ($NO_2$) but recent evidence on the association of $NO_2$ with a range of respiratory health effects now leads COMEAP to conclude that an affect remains after adjustments for other pollutants. COMEAP have not yet made a UK disease burden estimate for long term exposure to $NO_2$, but are working towards this, considering it sensible to regard $NO_2$ as causing some of the health impact observed in epidemiological studies. One meta-study they cite concluded that 'the magnitude of the effect of long-term

exposure to $NO_2$ on mortality is at least as important as that of $PM_{2.5}$' (Faustini *et al* 2014 cited in COMEAP 2015: p5).

The health effects of air quality are the principal reason for the establishment of air quality standards (ambient limit values). In the European Union, these are defined by Directive 2008/50/EC, with a $40 \mu g\,m^{-3}$ annual mean limit value for both $NO_2$ and $PM_{10}$. The UK compliance report (Defra 2013) indicates that in 2012 all 43 monitoring zones met the target for annual mean $PM_{10}$, but that 34 zones exceeded the limit value for annual mean $NO_2$, with four others compliant only due to a temporary margin of tolerance, to $48 \mu g\,m^{-3}$, granted under an extension. The UK Supreme Court ruled in 2015 that the government must develop a plan by the end of the year to further reduce $NO_2$ concentrations (Supreme Court of the UK 2015), whilst the European Commission (EC) have started legal proceedings for non-compliance with the Air Quality Directive (EU 2014). This may lead to substantive fines imposed on the UK, but the more substantive penalty is the loss of life that not meeting air quality objectives implies. Note that there is no lower threshold below which health effects are absent, hence compliance with an air quality standard does not imply no adverse health effect. Thus the World Health Organization (WHO) guideline value for annual average $PM_{10}$ is lower, at $20 \mu g\,m^{-3}$ ($40 \mu g\,m^{-3}$ for $NO_2$). In practice, regulatory standards are devised with reference to cost (net benefit) implications of compliance.

Poor air quality, and by implication its associated disease burden, is not distributed evenly in Britain. Geographically, it is largely an urban phenomenon, whilst socially, it is the poor who bear a disproportionate burden, as evidenced by national small-area analyses of GB (GB, the UK excluding N. Ireland) in 2001 (Mitchell and Dorling 2003, Walker *et al* 2003, Pye *et al* 2006). These analyses revealed that, of the 2.5 million people resident in areas where air quality breached the annual mean $NO_2$ limit value, over half were in the poorest 20% of the population. This is arguably the clearest 'environmental injustice' evidenced in the UK as air quality standards are intended to protect everyone. Furthermore, whilst the poor do contribute emissions (e.g. via older cars with higher emissions per km), they include those who emit least, and are most limited in their ability to avoid pollution, for example by moving home. Nevertheless, whilst air quality problems persist, a mix of technical, regulatory, and planning measures (see e.g. Defra 2007a) has led to major improvement in UK air quality since 2001. Here, we extend the 2001 environmental justice analysis of British air quality to 2011, to determine who has gained from a decade of air quality changes, and what this means for health inequalities and environmental justice.

## 1.2. Analysing environmental justice over time

Environmental justice (EJ) refers to the principle that people, regardless of socio-economic status or ethnicity, should fairly share the burdens of environmental hazards and the benefits of environmental amenities. EJ is often addressed in terms of fair distributions, of pollution or hazard, and procedural justice, which addresses whether people enjoy equal protection from hazards, and an equal opportunity to meaningfully affect decisions about their environment.

The EJ literature is large, and dominated by distributional studies. Initially, these were developed by activist groups in the USA, where EJ emerged from the civil rights movement, and came to prominence in 1982 following demonstration against siting of a toxic waste landfill in Warren County, North Carolina, a largely Afro-American community (Cutter 1995). Early studies often lacked rigour (Bowen 2002), but a wealth of evidence has since emerged to show that poor and minority communities are disproportionately exposed to a wide range of environmental hazards. For two decades EJ research was limited to the USA (Laurent 2011), with a focus on communities of colour, toxic waste facilities and industrial sites and emissions. EJ research in Europe began in the UK in the late 1990s, where deprivation was the primary social metric, and where the environment was conceived of in a broader way, with analysis of a greater range of 'bads' (industrial sites, landfills, air quality, flooding, road traffic accidents), as well as environmental 'goods', such as greenspace access (reviews in Lucas *et al* 2004 and Martuzzi *et al* 2010). EJ analyses subsequently developed across Europe including both Western (Lercher *et al* 2005, Chaix *et al* 2006, Laurian 2008, Laurian 2009, Fernández-Somoano *et al* 2013, Laurian and Funderburg 2013, Germani *et al* 2014, Padilla *et al* 2014), and Central and Eastern regions (Steger and Filcak 2008, Harper *et al* 2009, Branis and Linhartova 2012), and have most recently emerged from the Far East (Hedley *et al* 2008, Ma 2010, Schoolman and Ma 2012, Yasumoto 2013), the Middle East (Portman 2012), Australia and New Zealand (Pearce *et al* 2006a, Chakrobaty and Green 2014) and South America (Ramirez-Cuesta 2012).

Whilst the evidence base on environmental inequality is large and growing, its disparate nature hampers synthesis. This problem was evident in EJ analysis of air quality in Britain (reviewed by Mitchell and Dorling 2003) where analyses varied in geographic extent, spatial unit, social metric, atmospheric pollutant, and analytical method, and so precluded the drawing of any firm conclusion on the existence of environmental inequality until the national small area studies were developed. It is unsurprising then, that it is not yet possible to meaningfully synthesize findings on environmental inequalities for the field as a whole. However, the evidence on environmental inequality is sufficiently compelling that EJ enjoys widespread

public policy support (President 1994, UNECE 1998, HMG 2005: ch 6) in those territories where EJ has a longer history. However, claims of environmental *injustice* are often rather weak, as justice implies clear articulation of a value based normative element (what is a fair distribution?) and for many, an understanding of causality, that is, how unequal distributions have arisen.

Insight into processes producing environmental inequalities has been sought through longitudinal EJ studies (table 1). These mirror the development of the EJ field, in that they are dominated by US studies of hazardous sites, with a wider conception of the environment only tackled recently, by analysts outside the US, with few analyses of air quality. The studies test a range of theories (Liu 2001), most often addressing discriminatory siting of environmental hazards versus post-siting population dynamics. Some authors find evidence of overt discrimination in siting (Been 1994, Laurian and Funderburg 2013), and others that past discriminatory practice explains contemporary 'facially neutral' yet unequal siting, through designation of appropriate use standards and area restrictions on certain developments (Lord and Keaton 2010, Richardson *et al* 2010). Several authors refute any conclusions of biased siting and find post-siting population dynamics reflect those of the wider area (Oakes *et al* 1996), or that post siting housing market dynamics explain environmental inequalities (Anderson *et al* 1994, Hurley 1997, Mitchell *et al* 1999). Been (1994) found evidence that minority groups move into an area following hazard siting in urban areas, Boone *et al* (2014) that demographic and housing variables could not account for hazard siting, whilst others (Boone and Modarres 1999, Baden and Coursey 2002) concluded that location decisions by hazardous industries were based on availability of sparsely populated land with good accessibility, not area demographics.

Others find that minorities become over represented as housing and employment constrains their mobility more than other groups (Been and Gupta 1997, Stretesky and Hogan 1998, Mitchell *et al* 1999, Lercher *et al* 2005, Richardson *et al* 2010, Depro *et al* 2012, Ramirez-Cuesta 2012, Meir 2013, Pais *et al* 2014). Evidence is also found that poor and minority communities lack capacity for collective action to resist siting of hazardous activities (Hamilton 1993, Hamilton 1995, Hurley 1997, Pastor *et al* 2001, Saha and Mohai 2005, Laurian and Funderburg 2013) and conversely that middle class communities are better able to expel them (Ramirez-Cuesta 2012). The lack of ability to mobilize collective action is also linked to neighbourhood churning that limits the growth of a neighbourhood's social capital (Been 1994, Pastor *et al* 2001, Hipp and Lakon 2010). Flanquart *et al* (2013) explains the persistence of low income households near major hazard sites using cultural risk theory, with households trading off environmental risk for other benefits the area offers, or by

simply ignoring the risk. In a rare long-run analysis of an environmental good, Yasumoto (2013) concludes unequal provision of greenspace by private developers in Yokohama is a land market effect.

The methods used in unravelling the causes of environmental inequalities are varied. Statistical analysis of cross-sectional and time-series data feature prominently, but more recently textual analysis of planning meeting archives, and tracking mobility of individual households through real estate transaction records have added new perspectives (table 1). However, despite the methodological advances, and growing interest in temporal analyses, it is clear that no common understanding of the processes underlying evolution of environmental inequalities has emerged. This is a function of a small body of research, and a lack of systematic analysis that hampers comparison of studies, a problem that EJ research previously faced with respect to cross-sectional analyses (Bowen 2002, Kruize *et al* 2007). For example, for a common study area, changing the spatial unit of analysis caused apparent inequalities to disappear (Anderton *et al* 1994), whilst moving from a focus on proximity to exposure revealed inequalities not previously seen (Richardson *et al* 2010).

Developing an evidence base suitable for systematic evaluation of causal processes is evidently much more challenging than testing whether inequalities exist. The data constraints increase markedly as we go back in time, the passage of time increases the potential for multiple processes to drive inequality, and consideration must be given to analysis of institutional practices. These challenges may be seen as immaterial to those that argue that unequal is unfair. However, if environmental inequalities are to be avoided, or redressed effectively (e.g. by land use planning or procedural justice), then knowledge of how they change over time is a necessary prerequisite to understand their evolution. Given the complexity in evidencing the processes that explain environmental inequality, our aim here is necessarily cautious, and we do not seek to explain the changing social distribution of air quality. Rather, we seek to extend the UK's EJ evidence base, by first describing the temporal change in the social distribution of air quality, which in 2001 was arguably the clearest case of environmental injustice in the UK. Doing so allows us to discuss: (a) the implications for health inequalities in GB; (b) the compatibility of social justice and environmentally sustainability objectives; and (c) the extent to which progress towards EJ has been made in GB through air quality management.

## 2. Data and methods

Our analysis combines air quality and social data for GB for the population census years of 2001 and 2011. The air quality data is that which government returns

**Table 1.** Longitudinal environmental inequality studies.

| Author | Study area/spatial unit | Period | Environment metrics | Social metrics | Method and conclusion |
|---|---|---|---|---|---|
| Hamilton (1993, 1995) | USA/counties | 1987–1992 | Capacity added to 156 hazardous waste sites | Non-white, education, voter turnout, house values, income | Logistic regression modelling shows capacity increases are in areas with least social capacity for collective action; non-white population display lower collective action capacity. |
| Been (1994) | USA/various | 1970–1990 | Landfills and incinerators | Afro-American, income, home value | Descriptive statistics (three cross sections). Biased siting evident in rural areas. Urban inequity explained by housing market minority move in. Difference due to faster urban churning[a]. |
| Anderson *et al* (1994) | USA/census tracts | 1970–1990 | TSDFs[b] | Black/Hispanic; poverty; employment, house values | Descriptive statistics (three cross sections). Separate rural/urban analyses. TSDF siting suppresses house values but does not modify share of minority population. |
| Oakes *et al* (1996) | USA/census tracts | 1970–1990 | TSDFs | Black/Hispanic, poverty, unemployment | Descriptive statistics (three cross sections). No systematic race or class bias in TSDF siting. Demographic changes in host tracts mirror wider population. |
| Hurley (1997) | St Louis, USA/census tracts and site buffers | 1897–1984 | Abandoned toxic waste sites | Non-white population | Textual analysis of historical public/commercial siting records. Housing market dynamics and lack of political empowerment bring minority populations into area with hazardous facilities, but overt discrimination in siting decisions is absent. |
| Been and Gupta (1997) | USA/census tracts | 1970–1990 | TSDFs | Black/Hispanic, income, unemployment | Descriptive statistics (three cross sections) and logistic regression. Refutes market dynamics/minority move hypothesis; Hispanic and middle income communities more likely to gain waste sites, but not Afro-Americans. |
| Stretesky and Hogan (1998) | Florida, USA/census tracts | 1970–1990 | Superfund sites[c] | Black/Hispanic, poverty, pop density, unemployment | Descriptive statistics (three cross sections) and logistic regression. Higher exposure of ethnic groups from indirect discrimination, probably as reduced housing and employment choice. |
| Boone and Modarres (1999) | Commerce City, LA, USA | 1920s–1990s | Manufacturing sites with toxic emissions | Hispanics | Textual analysis. Current inequality is traced to 1920s decision to zone industrial expansion area, based on land availability and accessibility benefits to industry, not demographics. |
| Mitchell *et al* (1999) | S. Carolina, USA/counties | 1930–1990 | TRI facilities[d] | Black/White, family income | Descriptive statistics per decade. By 1990 clear inequity re Black communities produced by post siting demographics (e.g. housing market) not discriminatory siting. |

**Table 1.** (Continued.)

| Author | Study area/spatial unit | Period | Environment metrics | Social metrics | Method and conclusion |
|---|---|---|---|---|---|
| Pastor *et al* (2001) | LA County, USA/census tracts | 1970–1990 | TSDFs | Black/Hispanic, income, employment housing, pop density | Logistic regression. Minorities attract sites, sites do not attract minorities. Housing market dynamic not influential, but area demographic churning weakens social capital to resist siting. |
| Baden and Coursey (2002) | Chicago, USA/census tracts | 1960 and 1990 | Hazardous, solid waste, and waste generating sites | Black/Hispanic, income, pop density, proximity to road/waterway | Logistic and other regression. Sites locate in less populous accessible areas. Black communities not disproportionately exposed, but Hispanics are, following post siting move in. |
| Walker *et al* (2003) | England, census wards and km grid | 2001–2010 | $NO_2$ and $PM_{10}$ as annual average | Deprivation (2001 IMD[e] observations assumed for 2010) | Statistical comparison of modelled data. Inequality stable as air quality improves, except with exceedence of air quality standards, where poor benefit most from improvements. |
| Saha and Mohai (2005) | Michigan, USA | 1950–1990 | TSDFs | Black/White, family income, unemployed owner occupation <1 mile of TSDF site | Descriptive statistics per decade. Waste sites not sited in non-white neighbourhoods, but rise of environmental awareness results in new facilities being sited in communities of least political resistance, which are disproportionately non-white. |
| Mitchell (2005) | Leeds, UK/census wards a 100m grid | 1993–2015 | $NO_2$ for various road transport policies/futures | Deprivation (Townsend index)—static to 2015. | Statistical comparison using modelled air quality data. Measures that reduce aggregate $NO_2$, including road pricing and clean fuel vehicles, also reduce environmental inequality. |
| Lercher *et al* (2005) | Switzerland, rural areas | 1989–1994 | Traffic noise exposure and annoyance | Educational attainment | Descriptive statistics. Higher exposure amongst less educated develops, attributed to residential sorting as highly educated move to quieter areas. |
| Pye *et al* (2006) | UK/LSOA | 2003–2010 | $NO_2$, $PM_{10}$, $SO_2$ (2010 projected) | Deprivation. IMD 2004-5, static over time series | Descriptive statistics. Inequalities reduce slightly ($NO_2$) or do not change ($PM_{10}$, $SO_2$). Attributed to air quality policy. |
| Kruize *et al* (2007) | Rijnmond, Rotterdam, Netherlands | History before 2000–02 | Traffic noise, $NO_2$, hazard risk, green space | Income | Key informant interviews and text analysis. Environmental quality improves but environmental inequality occurs as no regulation to promote equality of access and exposure. |
| Echenique *et al* (2010) | SE and NE UK/model zones | 1997–2031 | $NO_X$, $PM_{10}$, noise, under regional spatial strategies | Income (modelled spatially in 5 yr steps) | Deterministic land use transport model linked to road transport emission model to test land-use and transport plan scenarios. As environmental quality improves inequality falls. |
| Hipp and Lakon (2010) | 6 counties in S. California/ census tracts | 1990–2000 | Toxicity weighted waste emission | Ethnic composition, education, income | Latent trajectory modelling. Toxic emissions fall, but more slowly in Latino and Asian communities where social churning results in weaker local collective action. |

**Table 1.** (Continued.)

| Author | Study area/spatial unit | Period | Environment metrics | Social metrics | Method and conclusion |
|---|---|---|---|---|---|
| Lord and Keaton (2010) | Baltimore, USA | 1940–2000 | Zoning decisions permitting dis-amenity uses | Race, income | Text analysis and emergence theory. Inequality not a post siting effect-1940s race segregation influenced zoning and hence later in appropriate use standards. Weakens over time. |
| Richardson et al (2010) | Scotland, bespoke zones | 1980–2001 | Exposure to IPPC[f] regulated landfills | Deprivation (Carstairs index) | Descriptive statistics. Higher exposure of poor due to planning bias pre-1980 then post-siting market dynamics. Multiple mechanisms operate, varying by place and time. |
| Depro et al (2012) | LA County, USA/census tracts | 1998–2008 | TRI sites | Income and ethnicity of households moving in study period (real estate Transaction records) | Structural equation modelling and Probit analysis. Residential sorting effect operates (Hispanics move in after other move out). Poor less likely to flee. Residential mobility undermines effort to restrict TRI sites in low income/ethnic areas. |
| Mitchell and Norman (2012) | England, LA districts | 1960–2007 | Tranquillity (Environmental intrusion index) | Deprivation (Townsend and Breadline Britain indices) | Descriptive statistics. Loss of tranquillity (due to infrastructure development) greatest in areas least deprived in 1960. Attributed to urban expansion and housing market dynamics. |
| Ramirez-Cuesta (2012) | Buenos Aries Argentina, Metropolitan Partidos | 1967–2008 | Industrial hazard density | Material deprivation, urban structure variables | Regression plus historical record analysis. Economic constraints dictate permanence of hazard sites; middle-class collective action helps expel hazards; local politics and lack of access to formal land market trap poor in hazard environment |
| Elliot and Frickel (2013) | Portland, USA/census tracts | 1950–2006 | Size, age and legacy of 2800 waste producing industries | Income, ethnicity, age, owner occupation, | Spatial and panel regression. Inequality evident in 2006 for active sites, but an equal pattern evident when relict (contaminated) sites included. Churning of industries dictates differential exposure, not demography or housing market. |
| Laurian and Funderburg (2013) | France, census communes | 1968–1999 | 107 Waste incinerators | Born abroad, foreigners, unemployed, population | Logistic econometric regression. Immigrants are predictors of incinerator location (elasticity 0.29). Most evident in 1970s when biased siting decision making operated, partly driven by unequal access to environmental decision making institutions. |
| Meir (2013) | LA county, USA, census tracts | 1998–2008 | Area index (TRI sites, crime, school quality) | Income, race, wealth | Hedonic regression. Low-income minorities, particularly Afro-American, less likely to receive large enough loans to buy a house in a cleaner area. |

**Table 1.** (Continued.)

| Author | Study area/spatial unit | Period | Environment metrics | Social metrics | Method and conclusion |
|---|---|---|---|---|---|
| Flanquart *et al* (2013) | Mardyck, France | 1990–2007 | Hazardous Seveso directive plants | Household size, age structure, housing tenure, education | Questionnaire asks why poor live near risky site. Economically constrained people, so catastrophic event risk is ignored and/or traded-off against pleasant area and community facilities. |
| Yasumoto *et al* (2014) | Yokohama, Japan, census tracts | 1988–2005 | Access to >1000 urban parks opened 1988-05 | Pensioners (>65), employed professionals | Descriptive statistics and logistic regression. No observed inequity in parks provided by city authority; parks provided by private developers favour areas developed for professional groups, and indicates a market driven park provision process. |
| Padilla *et al* (2014) | 4 cities in France/census blocks | 2002–2009 | $NO_2$ as an annual average | Employment, family type, immigrant, education, home tenure | Logistic regression. Inequality varies by city in strength and direction of association. Air quality improves with most benefit to poor of Paris and Marseilles, and least to poor of Lille and Lyon, probably a function of traffic density changes |
| Pais *et al* (2014) | USA/census tracts | 1991–2007 | TRI sites | Various, including race (Black/White) and income in a sample of 12 000 households | Latent class growth model of exposure trajectory probability. Black households more likely to experience persistent high exposure over life course. Inequality a racial income effect, and when controlling for income, a weaker racial only effect. |
| Boone *et al* (2014) | Baltimore, USA/neighbourhoods | 1960–2010 | Hazard density index based on TRI and other industry records | Race/ethnicity, education, income, housing tenure | Descriptive statistics and OLS regression. Association of hazard shifts from low income to White/Hispanic households. Hazard associated with low educational attainment throughout. Land dynamics and risk perception offer partial explanation. |

[a] Churning refers to succession, where one community or industry replaces another in a common area.

[b] TSDF—US EPA register of hazard waste Treatment, Storage and Disposal Facilities.

[c] Superfund sites are ex-waste and industrial sites in receipt of Federal support for site clean-up.

[d] TRI—US EPA Toxic Release Inventory.

[e] IMD—Index of Multiple Deprivation.

[f] IPPC—Integrated Pollution Prevention and Control sites are regulated by UK public authorities (according to type, size and administrative area).

to the EC under statutory reporting against the Air Quality Directive. These data are modelled by Ricardo-AEA Ltd under contract to the government (Defra), using the national atmospheric emissions inventory and a series of source and process specific models to produce an aggregate map of atmospheric concentration, calibrated and verified against a network of air quality monitoring stations (Stedman *et al* 2002, Brookes *et al* 2012). We analyse $NO_2$, where non-compliance problems persist, and fine particulates, a major public health concern evidenced by the disease burden calculated for the $PM_{2.5}$ fraction (COMEAP 2010, Gowers *et al* 2014). For temporal consistency, we use $PM_{10}$ data but note that much of the $PM_{10}$ mass is contributed by the $PM_{2.5}$ fraction. Annual average concentration values, presented on a 1 km grid for the nation are used.

Our social metric is deprivation, a state of disadvantage relative to the wider society in which people belong (Townsend 1987). People may be deprived of income, good quality housing, employment opportunities and environmental amenities (Dorling 1996). Since deprivation cannot be directly measured, schemes have been devised which use small-area input variables to construct composite indexes. There are a range of schemes in use in the UK (Jarman 1983, Townsend 1987, Carstairs and Morris 1989, Noble *et al* 2006), with others in use elsewhere (e.g. Bell *et al* 2007, Havard *et al* 2008, Norman *et al* 2015). Index choice is debated (Mackenzie *et al* 1998, Davey Smith *et al* 2001) but a high degree of correlation between schemes is found (Morris and Carstairs 1991, Hoare 2003). Deprivation measures influence the allocation of public resources (Simpson 1996, Blackman 2006) and are regularly used in models of outcomes, including health, in the UK (Boyle *et al* 2002, Diez Roux 2005, Norman *et al* 2005; Dibben *et al* 2006) and other countries (Lorant *et al* 2001, Tello *et al* 2005, Karpati *et al* 2006, Pearce *et al* 2006b).

We based our small area deprivation measure on the Townsend (1987) index which uses census variables on unemployment, non-home ownership, household overcrowding and non-car ownership, each of which is assumed to capture dimensions of deprivation (and which excludes an environmental quality measure which could introduce an auto-correlation problem in our analysis). These inputs are standardized to be on the same scale and have equal influence when combined into a single figure index. Deprivation indices capture deprivation on a cross-sectional basis, but the relative simplicity of the Townsend index and availability of the same information at different censuses means that, unusually, change in deprivation over time can also be calculated (Norman 2010a). To achieve this, the standardization process places areas relative to the average at national level of each input variable across time rather than just at one time point. Knowing whether areas have changed their level of deprivation over time can then be
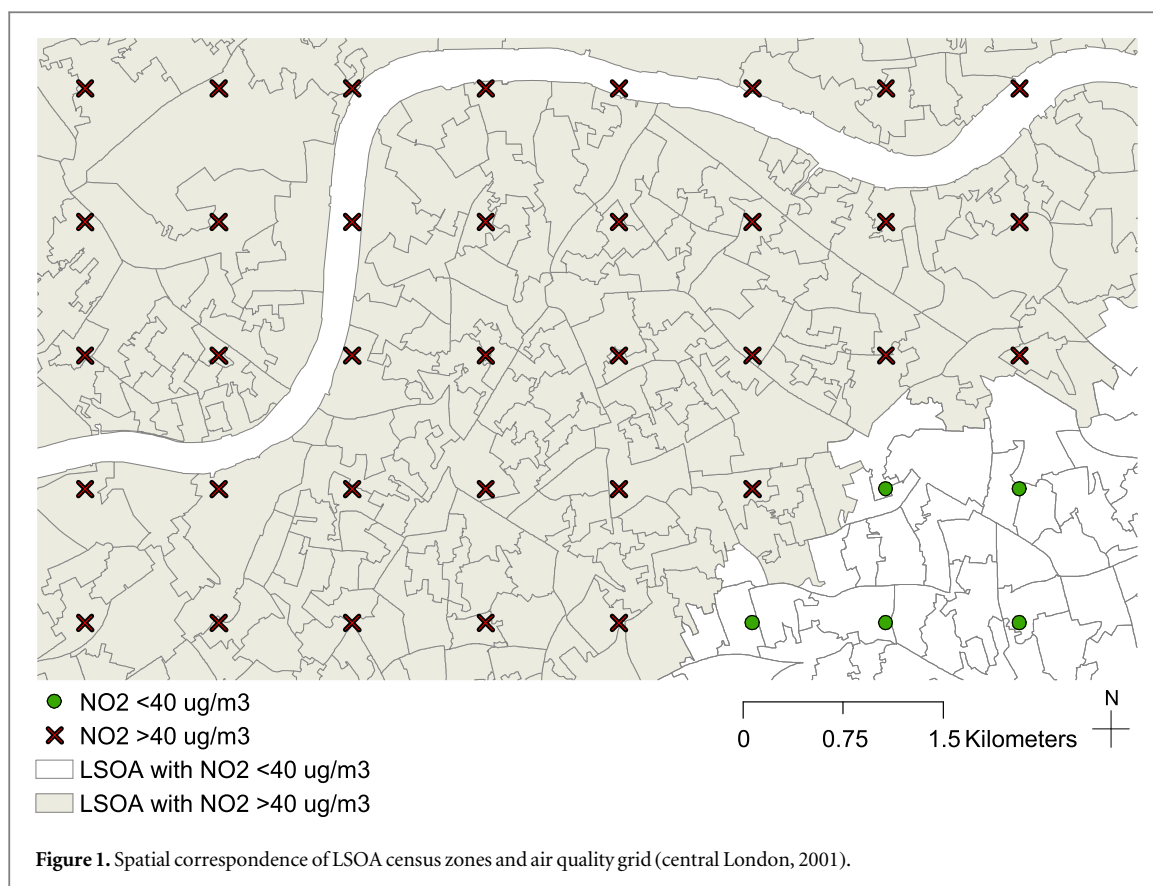
related to demographic change (Norman 2010b, Norman *et al* 2015), changes in health (Boyle *et al* 2004, Norman *et al* 2008, Exeter *et al* 2011) and changes in environmental quality (Mitchell and Norman 2012).

The environmental and social data were combined within a GIS to relate population, deprivation and air quality for each year. The UK has c. 42 000 census lower super output areas (LSOAs, known as Datazones in Scotland) each sized to have about 1500 people. Previous analyses (Mitchell and Dorling 2003, Walker *et al* 2003) were conducted using census wards (c. 8500 in GB), hence the LSOA analysis represents a more spatially resolved demography. However, because population density varies, sparsely populated LSOAs will be large and so capture many air quality grid points (where we calculate the average), but others may be very small and capture none. The latter is particularly the case in dense urban areas (figure 1). For LSOAs that had no incident air quality grid point, values were determined as the non-area weighted mean value of all kilometre grid cells overlapping the LSOA in question. Data were then analysed by ranking all LSOAs by deprivation status, and then sub-dividing them into equal population deciles, decile 1 (D1) being least deprived. Deciles have 5.71 million people in 2001, and 6.13 million in 2011. Air quality statistics were then calculated for each decile. Statistically, this is a very simple analysis yet it is powerful, as it does not deal with sample data, but the entire population, based on small area data. Thus the descriptive statistics reported below, are an excellent representation of the changing social distribution of air quality in GB.

## 3. Results

### 3.1. The social distribution of annual average pollutant concentrations

Table 2 summarizes $NO_2$ and $PM_{10}$ concentrations in 2001 and 2011. Nationally, $NO_2$ concentrations decline reflecting a reduction in emission, particularly from the road transport sector as a result of tighter Euro emission standards for new vehicles (Bush *et al* 2014). There is an 88% fall in the number of grid cells where annual average $NO_2$ concentrations exceed the 40 $\mu$g m$^{-3}$ limit value, but a significant degree of non-compliance remains hence the current EC legal action. Conversely, there are no exceedances of the 40 $\mu$g m$^{-3}$ annual average standard for fine particulates, but the national mean concentration has increased by 7%, with a substantial increase in the number of grid cells above the WHO 20 $\mu$g m$^{-3}$ annual average guideline value. This is inconsistent with the National Atmospheric Emission Inventory which shows that primary particulate emissions decline during this period. Harrison *et al* (2008) identify several possible explanations for this, including that air masses originating over the European mainland could be more prevalent, 'importing'

**Figure 1.** Spatial correspondence of LSOA census zones and air quality grid (central London, 2001).

**Table 2.** UK air quality, 2001–2011.

| Metric[a][b] | | 2001 | 2011 |
|---|---|---|---|
| Nitrogen dioxide (N0$_2$) | Mean value ($\mu$g m$^{-3}$) | 12.61 | 7.49 |
| | Standard deviation ($\mu$g m$^{-3}$) | 8.88 | 5.30 |
| | Maximum ($\mu$g m$^{-3}$) | 95.20 | 56.20 |
| | Number of km$^2$ grid cells >40 $\mu$g m$^{-3}$ | 761 | 86 |
| Fine particulates (PM$_{10}$) | Mean value ($\mu$g m$^{-3}$) | 12.14 | 13.00 |
| | Standard deviation ($\mu$g m$^{-3}$) | 2.50 | 3.80 |
| | Maximum ($\mu$g m$^{-3}$) | 29.54 | 27.77 |
| | Number of km$^2$ grid cells >40 $\mu$g m$^{-3}$ [c][d] | 0 | 0 |
| | Number of km$^2$ grid cells >20 $\mu$g m$^{-3}$ [d] | 386 | 3802 |

[a] Concentration values are based on annual averages.

[b] For NO$_2$ $n = 244\,938$ grid cells, and for PM$_{10}$, $n = 244\,374$ (no data for Isle of Man). The original 2011 data set is larger as the model extends beyond the coast, but points over the sea were excluded.
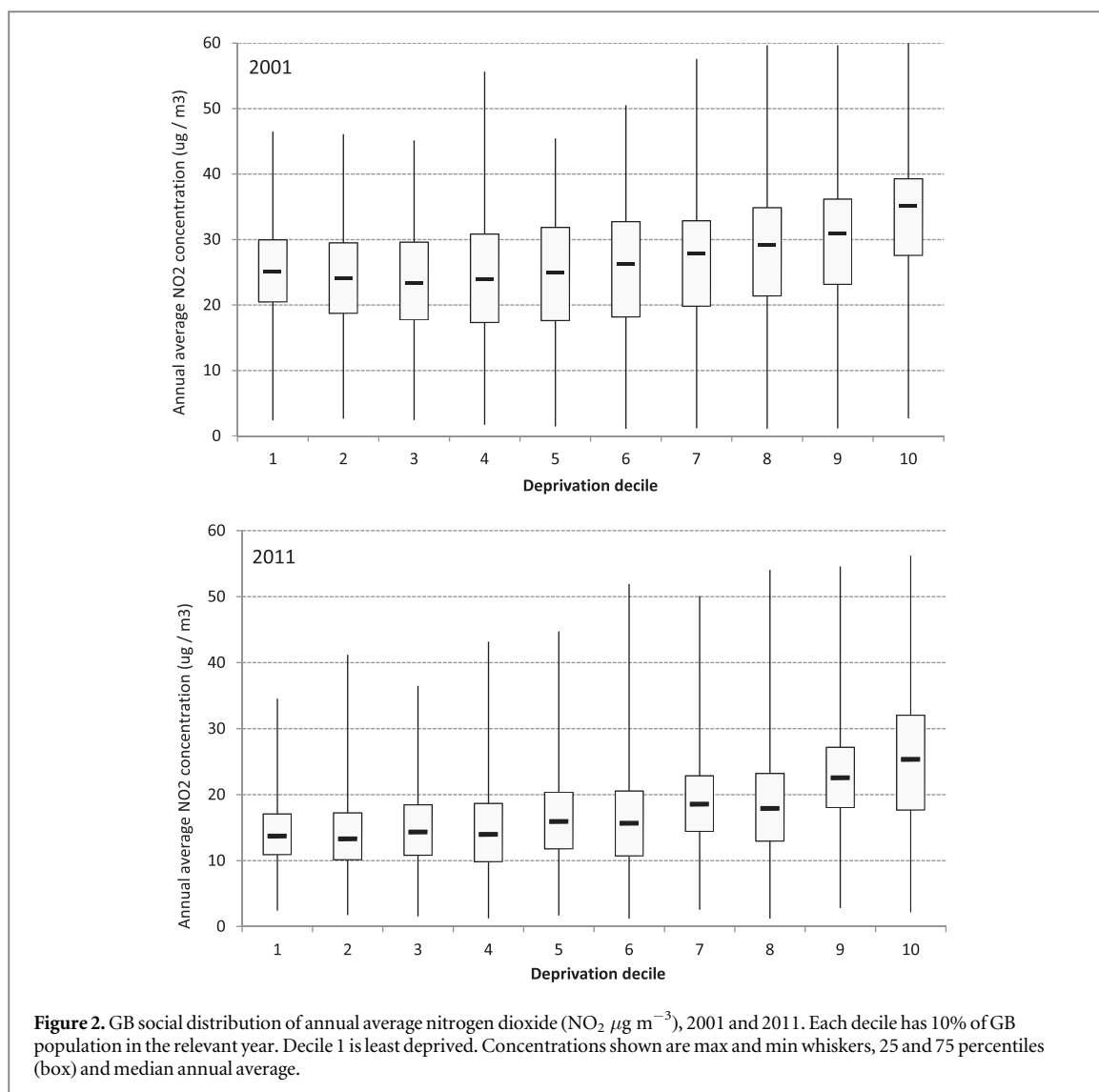
[c] EC limit value.

[d] World Health Organization (WHO) guideline value.

particulates to the UK. Spring 2011 did have a higher degree of secondary particulates than normal (Bush *et al* 2014), but based on assessment of multi-year trend data, Harrison *et al* discount air mass trajectory as an explanation for the lack of expected decline in particulate concentration.

More viable explanations relate to inaccuracies in the emission inventory, including underestimation of emissions from wood combustion and residential heating, greater non-exhaust emissions (tyre and brake wear and particulate resuspension) from increasing traffic volumes, and cycle beating, whereby vehicle manufacturers make adjustments in the emissions test, such that reported emissions are lower than those that occur in real world driving (Harrison *et al* 2008). From analysis of all emission sources, Fuller and Green (2006) concluded that road transport is the cause of rising particulate concentrations observed for London, and it is reasonable to assume this applies to other urban centres. The rise in popularity of diesel cars (due to lower CO$_2$ emission and associated tax) is likely the key factor behind the increase in particulate concentrations, especially given the recent revelation that a European volume car

**Figure 2.** GB social distribution of annual average nitrogen dioxide (NO$_2$ $\mu$g m$^{-3}$), 2001 and 2011. Each decile has 10% of GB population in the relevant year. Decile 1 is least deprived. Concentrations shown are max and min whiskers, 25 and 75 percentiles (box) and median annual average.
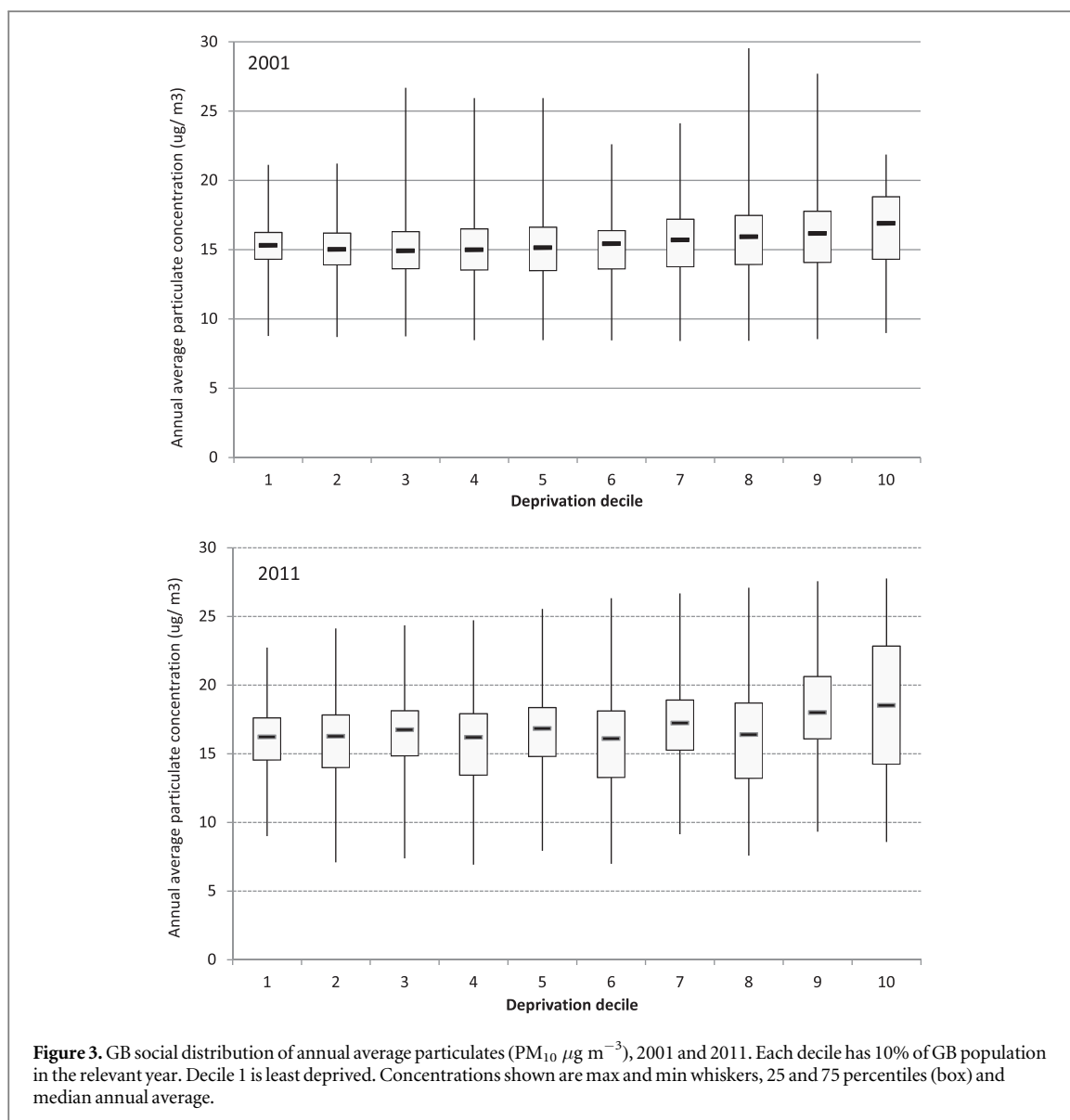
manufacturer has since used a software device to under report emissions recorded in official emission tests in the USA (US EPA 2015). In 2001 there were 25.1 million cars in use in GB, of which 13.8% were diesel, rising to 30.8% (of 28.4 million cars) in 2011 (DfT 2015). This trend is also of concern with respect to NO$_2$, as real world emission testing has shown that new diesel vehicles emit about seven times the NO$_X$ required by the Euro 6 emission standard, and so diesel's popularity will exacerbate difficulties in meeting NO$_2$ air quality limit values (Franco *et al* 2014). False emission factors for diesel vehicles may also explain why a larger model calibration factor is needed in the integrated PM$_{10}$ model for 2011, relative to 2001, to force a fit between modelled and (higher) observed concentration data.

Figure 2 illustrates the social distribution of NO$_2$. The 2001 pattern is consistent with prior studies (Mitchell and Dorling 2003, Walker *et al* 2003, Pye *et al* 2006) and shows a steady increase in NO$_2$ concentration as deprivation increases, with the most deprived decile (D10) experiencing a median annual

average NO$_2$ concentration of 35.1 $\mu$g m$^{-3}$, compared to 25.2 $\mu$g m$^{-3}$ for the least deprived D1 (and the minima of 23.4 $\mu$g m$^{-3}$ experienced by D3). Thus the most deprived areas experience NO$_2$ concentrations that are 40% higher than those of the most affluent. The slight upward tail from D3 to D1 is interpreted as the more affluent trading off the best air quality for the benefits of a good urban location (Mitchell and Dorling 2003). All deprivation deciles experience breaches of the 40 $\mu$g m$^{-3}$ annual average standard, although the more deprived experience more extreme exceedances.

By 2011, all groups experience a reduction in NO$_2$ concentration. The slight upward tail for the most affluent (D1) is greatly reduced and is very close to the minimum experienced by any group (D2). A clear social gradient in NO$_2$ persists, with annual average concentrations experienced by the most deprived (D10) now 85% higher than that of the least deprived (D1). Maximum values have fallen across all social groups, and some of the more affluent groups (D1, D3) no longer experience NO$_2$ concentrations in

**Figure 3.** GB social distribution of annual average particulates (PM$_{10}$ $\mu$g m$^{-3}$), 2001 and 2011. Each decile has 10% of GB population in the relevant year. Decile 1 is least deprived. Concentrations shown are max and min whiskers, 25 and 75 percentiles (box) and median annual average.

breach of the 40 $\mu$g m$^{-3}$ annual limit. These data suggest that NO$_2$ has improved substantially for all social groups, with the greatest improvements in the more affluent areas.

Figure 3 illustrates the social distribution of PM$_{10}$. As with NO$_2$, a social gradient in concentration is evident in 2001 and 2011. Annual average concentrations are highest in the most deprived group (D10), 10.5% above that of the least deprived (D1) group in 2001, rising to 14.2% in 2011. Although there are no exceedances of the 40 $\mu$g m$^{-3}$ annual average standard, maximum values have not fallen as they have done with NO$_2$, and a rising trend in maximum PM$_{10}$ values with increasing deprivation is evident, especially in 2011. This suggests that exceedances of the 50 $\mu$g m$^{-3}$ 24 h limit value, which do occur, are likely to be more common in areas of higher deprivation.

Table 3 further illustrates the changes in air quality by social group over the decade. These data show a clear pattern with the least deprived experiencing a greater share of improvements in air quality (NO$_2$),

and the most deprived experiencing a greater share of declines in air quality (PM$_{10}$).

## 3.2. The social distribution of air quality standards' exceedance

Next we analyse the changing social distribution of non-compliance with the NO$_2$ annual mean limit value (there are no equivalent exceedances for PM$_{10}$). The major improvement in air quality over the decade is readily seen (figure 4). In 2001 2.68 million people (4.7% of the population at the time) lived in a LSOA that failed the EC Directive limit value, falling to 0.61 million in 2011 (1.0% of the population). Thus the chance of living in a non-compliant area fell from around one in twenty to one in a hundred. However, a very strong and steepening social gradient is also evident (nb. the log scale). In 2001 66% of people in an exceedance area were from the most deprived population quintile (Q5), rising to 85% in 2011. The Q5:Q1 ratio is 15 in 2001, but rises to 892 in 2011, as

**Table 3.** Change (%) in annual average air quality statistic by social group, 2001–2011.

| NO₂ | Equal population deprivation decile[a] | | | | | | | | | |
|---|---|---|---|---|---|---|---|---|---|---|
| | 1 | 2 | 3 | 4 | 5 | 6 | 7 | 8 | 9 | 10 |
| Mean | −44.3[b] | −43.2 | −37.6 | −40.1 | −33.2 | −37.0 | −28.5 | −33.8 | −22.2 | −24.0 |
| Median | −45.4 | −44.5 | −38.5 | −41.6 | −36.1 | −40.5 | −33.5 | −38.5 | −27.0 | −27.7 |
| 75%ile | −43.1 | −41.6 | −37.6 | −39.4 | −36.0 | −37.2 | −30.5 | −33.5 | −24.9 | −18.4 |
| Maximum | −25.7 | −10.6 | −19.1 | −22.4 | −1.5 | 2.8 | −13.0 | −9.4 | −8.5 | −7.3 |

| PM₁₀ | Equal population deprivation decile | | | | | | | | | |
|---|---|---|---|---|---|---|---|---|---|---|
| | 1 | 2 | 3 | 4 | 5 | 6 | 7 | 8 | 9 | 10 |
| Mean | 5.4 | 6.0 | 10.6 | 5.5 | 11.1 | 4.4 | 12.0 | 4.5 | 15.4 | 13.1 |
| Median | 6.0 | 8.3 | 12.2 | 8.0 | 11.2 | 4.4 | 9.8 | 2.9 | 11.3 | 9.6 |
| 75%ile | 8.4 | 10.1 | 11.2 | 8.6 | 10.4 | 10.6 | 10.0 | 7.1 | 16.0 | 21.3 |
| Maximum | 7.6 | 13.8 | −8.7 | −4.7 | −1.5 | 16.5 | 10.6 | −8.3 | −0.5 | 27.0 |

[a] Population decile 1 is least deprived, 10 most deprived.

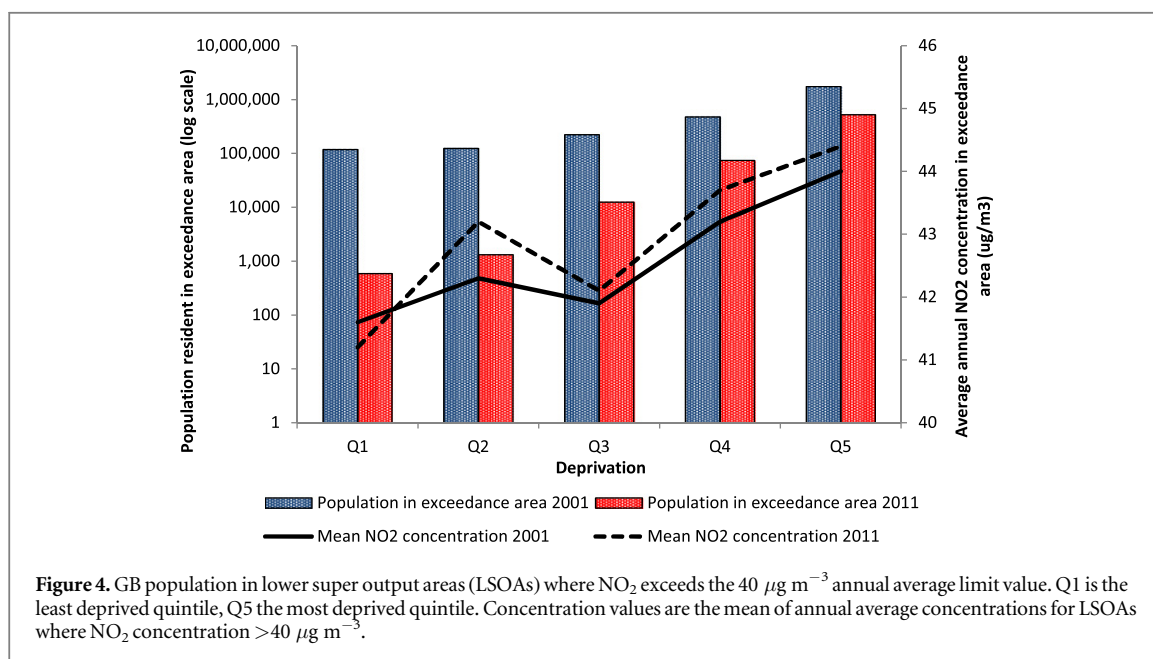[b] A negative value indicates improvement in air quality, 2001–2011.



**Figure 4.** GB population in lower super output areas (LSOAs) where NO₂ exceeds the 40 $\mu$g m$^{-3}$ annual average limit value. Q1 is the least deprived quintile, Q5 the most deprived quintile. Concentration values are the mean of annual average concentrations for LSOAs where NO₂ concentration >40 $\mu$g m$^{-3}$.

**Table 4.** GB population resident in LSOAs which exceeded the 40 $\mu$g m$^{-3}$ NO₂ annual average standard in 2001 but not 2011.

| | Population weighted deprivation quintile | | | | |
|---|---|---|---|---|---|
| | Q1 | Q2 | Q3 | Q4 | Q5 |
| Population in LSOAs non-compliant in 2001, but compliant in 2011 | 117 243 | 122 836 | 210 659 | 400 833 | 122 5450 |
| Population in LSOAs non-compliant in 2001, but compliant in 2011 (% of 2001 population in non-compliant LSOAs) | 99.5 | 98.9 | 94.4 | 84.4 | 70.0 |

exceedances of the NO₂ limit are disproportionately eliminated from the least deprived areas. There is also a social gradient in the concentration values, with the more deprived areas experiencing exceedances of the limit value by a greater margin than the least deprived. This is significant as higher concentrations are associated with higher disease burden.

The air quality improvement over the decade delivered a 77% reduction in the number of people exposed to air quality that breached the annual average NO₂ limit value, some 2.08 million people, but this benefit is not distributed evenly. Table 4 shows that of the least deprived (Q1) people resident in an LSOA that did not comply with the air quality standard in 2001, the LSOA of virtually all of them (99.5%) was compliant by 2011. The corresponding figure for the most deprived quintile (Q5) is 70%. That is, there is a social gradient in those LSOAs changing from air quality failure to compliance. A few LSOAs move in the opposite direction, from compliance in 2001 to non-

**Table 5.** GB population resident in LSOAs compliant with the 40 $\mu$g m$^{-3}$ NO$_2$ annual average standard in 2001 but not 2011.

| | Population weighted deprivation quintile | | | | |
|---|---|---|---|---|---|
| | Q1 | Q2 | Q3 | Q4 | Q5 |
| LSOAs affected | 1 | 1 | 5 | 2 | 19 |
| Total population | 540 | 1040 | 7310 | 2859 | 21 228 |
| Mean NO$_2$, increase 2001–11 ($\mu$g m$^{-3}$) | 9.6 | 5.0 | 9.6 | 4.0 | 5.8 |

compliance in 2011. These are in central London (Tower Hamlets, Hammersmith and Fulham), Southampton, and Cardiff. A small total population is affected (33 000 people), but a strong social gradient exists here too, with two thirds of the population in areas that become non-compliant in the most deprived quintile (table 5). These data show that whilst there has been substantial improvement in compliance with air quality directive standards over the decade, the poor have not benefited to the same degree as the affluent, and where air quality has declined, it has largely been in poor neighbourhoods.

There are no breaches of the EC directive annual average limit value for PM$_{10}$, but the number of people who live in areas where concentrations are above the 20 $\mu$g m$^{-3}$ WHO guideline value has increased markedly from 2001–2011 (table 6). Of these 70% were in the most deprived quintile in 2001, falling to 59% in 2011.

# 4. Discussion

## 4.1. Introduction
EJ studies increasingly track change in environmental inequality over time to better understand what causes the inequality. For many, understanding causation is important in judging the fairness of observed distributions, and whether, and what, interventions are required to redress perceived injustice. There is however, a lack of any systematic approach to such studies which collectively address a mix of environmental and social metrics, places, and time and space scales, with environmental inequality variously attributed to market dynamics, residential sorting, planning policy and discrimination (table 1). It is likely that multiple processes operate. For example, Richardson *et al* (2010) concluded that hazardous waste disposal siting in Scotland is a function of pre-1980 planning bias and the subsequent post siting market dynamics, with deprived areas gaining economically, but more slowly than others, constraining relocation of deprived households away from hazards.

Few temporal EJ studies have addressed air quality directly. Padilla *et al* (2014) investigated NO$_2$ in four French cities over a seven year period and found that

the poor gained most, or least, depending upon the city and its road traffic density. Kruize *et al* (2007) concluded for Rotterdam that NO$_2$ concentrations had fallen, but least for low income groups. Conversely, other temporal studies have indicated that as air quality improves, inequality declines (Mitchell 2005, Walker *et al* 2003, Pye *et al* 2006, Echenique *et al* 2010), but these studies are limited as the social metrics are static over time, or the air quality is a forecast, or both. Furthermore most air quality studies are of individual cities, hence prone to the ecological fallacy (Miao *et al* 2015) that makes drawing general conclusions on the dynamics of environmental quality and inequality challenging. Our study is the first small area-national study of the social distribution of air quality over time, and has the advantage of using observed data (directly, or in calibration of the necessary spatial model). Further work is needed to understand what explains our observations (e.g. spatio-temporal analysis of different pollutant sources, investigation of changing patterns of deprivation), but we are able to draw conclusions on who has benefited from air quality change in GB over a decade of air quality management, which allows us to consider wider questions on health inequalities and EJ.

## 4.2. Implications for health inequalities
COMEAP's estimate of 29 000 premature deaths is a statistical means of conveying the total disease burden due to NO$_2$ (in 2008). In practice this a loss of life across the entire population, including in areas that comply with limit values, as the NO$_2$ (and PM$_{10}$) exposure-response relationship has no lower (no effect) threshold. Thus where air quality improves, disease burden should fall. The unequal social distribution of air quality in 2001 was substantial, implying a higher disease burden amongst the poor. Indeed, Wheeler and Ben-Shlomo (2005) in a study of England in 1996, observed an association of decreased lung function with poor air quality remained after confounders, such as social class, had been adjusted for. Air quality changed significantly 2001–2011 so *ceteris paribus* total disease burden will also have changed, but because air quality does not improve equally for all, we assume that change in disease burden will also be unequal. Everyone benefits from improving NO$_2$, but the poor do so least of all, whilst PM$_{10}$ increases are more numerous in poor areas. Therefore we conclude that air quality will be a more important factor in explaining air quality related health inequalities observed in 2011 than in 2001. Understanding these health inequalities does however require further work to understand the effects of the poor's lower base respiratory health and access to health services (Defra 2007b, p 201) and the role of daily mobility patterns in population exposure.

**Table 6.** GB population in LSOAs where $PM_{10}$ exceeds WHO 20 $\mu g\ m^{-3}$ annual average guideline.

| | | Population weighted deprivation quintile | | | | |
|---|---|---|---|---|---|---|
| | | Q1 | Q2 | Q3 | Q4 | Q5 |
| 2001 | Population in exceedance area | 22 105 | 32 926 | 47 670 | 109 545 | 506 739 |
| | Mean $PM_{10}$ concentration $\mu g\ m^{-3}$ [b] | 20.80 | 21.57 | 21.27 | 21.34 | 20.71 |
| 2011 | Population in exceedance area | 200 974 | 520 439 | 975 914 | 2 054 249 | 5 410 058 |
| | Mean $PM_{10}$ concentration $\mu g\ m^{-3}$ [b] | 20.80 | 20.85 | 21.16 | 21.40 | 22.77 |

[a] Q1 is the least deprived quintile, Q5 the most deprived quintile.

[b] Concentration values are the mean of annual average concentrations for LSOAs where $NO_2$ concentration $>40\ \mu g\ m^{-3}$.

## 4.3. Implications for achieving environmental justice

From 2001–2011 environmental inequality increased, but environmental quality improved substantially, with over two million people no longer exposed to $NO_2$ in breach of the EC limit value. From an environmental justice perspective should this be viewed as a failure or a success? This depends upon the conception of justice adopted. A utilitarian perspective would view the changes favourably, as although benefits are unequal, total public welfare has increased (unless the health gains from lower $NO_2$ are offset by losses from higher $PM_{10}$). If this total welfare increase comes with none suffering a loss of welfare (a Pareto improvement) then this is a just outcome. If raising total welfare results in some experiencing a loss of welfare, then justice can be restored if compensation is made (Kaldor 1939, Hicks 1939). Most enjoy improving air quality 2001–2011, but some, including those in central London suffer a decline. However, compensation to deliver EJ would only be appropriate if the reductions in air quality were a consequence of air quality management aimed at raising overall welfare. This is quite plausible (e.g. London's congestion charge zone may redistribute pollution) but requires analysis to understand the causal process behind the observed deterioration in air quality.

Conversely, a radical egalitarian view would see air quality management in GB as a failure, as benefits of air quality improvement have not been distributed equally. From a contractarian perspective we would likely draw a similar conclusion, but for a different reason than a strict demand for equality. Air quality standards are one expression of the social contract between state and citizen, and are set blind to the characteristics of individuals, behind a 'veil of ignorance' (Rawls 1971). The standards are a right intended to offer equal protection to all. However, both compliance, and rate of change of compliance, is strongly biased in favour of the affluent. Thus whilst inequality in annual average concentrations below the standard could be viewed as just, the application of a standard implies that air quality change 2001–2011 has been unjust.

An alternative, libertarian, conception of justice is based not on the observed distribution, but on how it arose. Nozick's (1974) entitlement theory sees a distribution as just if the possessions involved are fairly acquired (e.g. via work on common property) or justly transferred (bought, gifted). Social inequality in GB's air quality can thus be viewed as just, so long as the processes that produce those inequalities adhere to Nozick's rules of just acquisition. For example, these rules can be applied to the housing market, an important factor in determining the geographical distribution of people, and hence deprivation. In this perspective, process takes precedence over outcome, and hence an understanding of causality is particularly important in judging fairness. Thus judging whether GBs air quality change has been just is not simple. It implies adoption of a normative position, and in some cases, investigation to understand the processes that give rise to the observed distributions.

## 4.4. Implications for compatibility of social justice and environmental sustainability objectives

Our analysis offers evidence to test a claim in the EJ literature that social justice and environmental sustainability are not always compatible objectives. Dobson (2003) observes how EJ advocates and policy makers assume the two are compatible, such that action to progress one will progress the other, but notes very little theory or empirical evidence has been developed to test this. The incompatibility conclusion arises as the justice and sustainability movements have different strategic objectives. Dobson gives the example of waste disposal, where justice might involve redistributing landfills to more affluent communities, in contrast to environmental sustainability where total waste produced would be reduced. Dobson notes how testing compatibility empirically would require environmental sustainability and social justice concepts to be more clearly defined. That is, what aspect of the environment is to be sustained, and what conception of justice adopted? Empirical research would then be needed across a full range of such meanings to demonstrate compatibility. Our study contributes to this research agenda: we have one very clear conception of environmental sustainability (compliance with air quality limit values, which we can also track over time) which is associated with a clear contractarian conception of justice (where all have the right to enjoy air quality that meets publicly agreed standards). Our

analysis shows that whilst compliance rates have improved (progress towards sustainability), the remaining non-compliance is increasingly biased towards the poor. This supports Dobson's (2003, p 83) 'reluctant conclusion that social justice and environmental sustainability are not always compatible objectives'. The current legal proceedings may lead to compliant air quality in future, but this will have been achieved through a drive to meet environmental sustainability objectives, and not those related to a just distribution.

### 4.5. Implications for policy and process

Our analysis addresses distributional justice, but environmental justice also encompasses procedural justice and the recognition of all people in decision making (Walker 2009). Air quality management in the UK is a highly regulated process, and it could be argued that this essentially top down process already offers procedural justice. After all, although air quality is not wholly compliant, the EU prosecution, and UK Supreme Court ruling represent serious efforts to ensure air quality complies with standards everywhere, and by extension, for everyone. However, an alternative conception of procedural justice argues that the enforcement of minimum air quality standards approach is problematic as: standards do not specifically recognize vulnerable populations, those most affected by non-compliant air quality have little say in the setting of standards or their enforcement, there is no safe level for some pollutants, there is no consideration of responsibility for emissions, and the limits of science and monitoring mean there are significant uncertainties over the air quality people actually experience (Walker 2012).

For these reasons, community environmental justice activists argue for procedural justice in which there is a more open, inclusive and deliberative decision making process that allows those most at risk to be involved in scrutinizing scientific information, setting standards that better reflect needs of vulnerable groups, and deciding on interventions to tackle air quality problems. An important first step here would be for public authorities to provide air quality information by population group, and how this changes in response to underlying trends and local plans and projects. The UK has several mechanisms that should act to drive this, including the 2010 Equality Act (although note that low income, the focus of our analysis, is not a protected characteristic), the statutory Strategic Environmental Assessment process that is routinely broadened to include wider sustainability concerns, and 'The Green Book' (HM Treasury 2014) that requires publicly funded bodies to analyse policies, programmes and projects for public value and risk, and which includes guidance on distributional analysis. Despite these drivers, equity does not feature

in the statutory Local Air Quality Management (LAQM) process or even critical evaluations of it (IHPC 2010, Moorcroft and Dore 2013).

Air quality management in the UK remains a compliance focussed process. Where air quality is not compliant, LAQM empowers local authorities to declare an air quality management area (AQMA) (around 60% of local authorities have done so), and develop an associated air quality action plan (AQAP) (see Defra 2015 for details of current AQAP proposals). AQMAs tend to have above average deprivation (Gegisian *et al* 2006). Unfortunately, the lack of attention to social characteristics in LAQM means that the environmental justice implications of an AQAP are neglected. However, there is a bigger problem here. That is, that whilst the air quality review and assessment part of the LAQM process has worked reasonably well, the action planning process has not, with air quality improvements falling well short of that required to achieve compliance (Moorcroft and Dore 2013).

Barnes *et al* (2014) identify a range of barriers to effective LAQM including issues of resourcing and agency co-ordination, but conclude that the principal barrier is flawed subsidiarity. That is, local authorities have the responsibility to improve air quality, but not the power to do so. In large part this is because poor air quality in the UK is a result of national growth in road transport, which is largely beyond the control of local authorities. Central government policy and funding is needed to manage traffic demand. Barnes *et al* (2014) thus conclude that responsibility for AQAPs should be removed from local authority environmental health departments and integrated into cross departmental policy at a strategic level, with the local strategies developed in conjunction with a similar cross-departmental team in central government. This would overcome barriers and conflicts between departments (with environment, transport, health, and climate change responsibilities) that currently operate with insufficient integration. The implication is that any injustice in air quality cannot be tackled purely at the local level in the UK, but engagement at multiple-levels of government, and across multiple interests, not just environment, is needed. This may prove challenging for environmental justice communities more familiar with dealing with local, single issue problems.

## 5. Conclusion

Our analysis shows that improvement in GB's air quality has been substantial but unequal. Annual average $NO_2$ concentrations have fallen markedly, but the rate of improvement has been slower for the more deprived. Conversely annual average $PM_{10}$ concentrations have risen, and done so more quickly for the poor. For exceedance of the $NO_2$ EC directive limit

value effectively all of the most well off are lifted out of exceedance, compared to 70% of the most deprived. Of the millions who become exposed to $PM_{10}$ concentrations above the WHO annual guideline value, 58% are in the most deprived 20% of the population. We anticipate that disease burden due to $NO_2$ will have declined since 2001, increased for $PM_{10}$, and that air quality related health inequalities will have grown 2001–2011. Whether GB's poor air quality is now more fairly distributed depends upon the justice conception adopted. A utilitarian perspective suggests it is, a libertarian perspective that it could be (depending on causality), and an egalitarian perspective that it is not. The social contract perspective, arguably dominant in Britain, suggests that EJ will not be achieved until full compliance with environmental legislation is achieved. This echoes Chaix *et al* (2006) who concluded from their analysis of air quality in Sweden, a country noted for its egalitarian welfare system, that stronger environmental enforcement was needed to deliver EJ. Regardless of which justice conception is adopted, access to information on the changing social distribution of air quality would better support procedural justice with respect to air quality. Finally, we conclude that our study supports Andrew Dobson's 'reluctant conclusion', that environmental sustainability and social justice are not always compatible objectives.

## Acknowledgments

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