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An environmental justice analysis of British air quality

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Abstract. This paper presents the results of the first national study of air quality in Britain to consider the implications of its distribution across over ten thousand local communities in terms of potential environmental injustice. We consider the recent history of the environmental justice debate in Britain, Europe, and the USA and, in the light of this, estimate how one aspect of air pollution, nitrogen dioxide (NO_2) levels, affects different population groups differentially across Britain. We also estimate the extent to which people living in each community in Britain contribute towards this pollution, with the aid of information on the characteristics of the vehicles they own. We find that, although community NO_x emission and ambient NO_2 concentration are strongly related, the communities that have access to fewest cars tend to suffer from the highest levels of air pollution, whereas those in which car ownership is greatest enjoy the cleanest air. Pollution is most concentrated in areas where young children and their parents are more likely to live and least concentrated in areas to which the elderly tend to migrate. Those communities that are most polluted and which also emit the least pollution tend to be amongst the poorest in Britain. There is therefore evidence of environmental injustice in the distribution and production of poor air quality in Britain. However, the spatial distribution of those who produce and receive most of that pollution have to be considered simultaneously to see this injustice clearly.

Introduction

Environmental protection and social justice, two of the fundamental tenets of sustainable development, are brought together by 'environmental justice' (EJ), a concept of growing interest to researchers and policymakers. Cutter (1995) defines EJ as equal access to a clean environment and equal protection from possible environmental harm irrespective of race, income, class, or any other differentiating feature of socioeconomic status. Thus EJ research seeks to determine whether marginal and/or minority groups bear a disproportionate burden of environmental problems, and whether planning policy and practice affecting the environment are equitable and fair. Through a national ward-level analysis, in this paper we investigate the equity implications of air quality, as measured by nitrogen dioxide (NO_2) levels, in Britain. In doing so, we aim to add to the small but growing body of work on EJ in the United Kingdom, and hope to assist those who have responsibilities under the National Air Quality Strategy (NAQS) (DETR, 2000) for delivering fair and effective air-quality management policies.

Environmental justice

The EJ approach was pioneered in the USA by civil rights activists concerned that landfills and polluting industries were invariably sited within predominantly black communities or indigenous peoples' reservations. Although the US academic literature did much to document the nature and extent of environmental inequality (Bullard, 1990; Lavelle and Coyle, 1992), class actions brought against civil authorities on the grounds of unjust planning decisions have been largely unsuccessful. Bowen's (2002) review of the US EJ literature since its beginnings in the early 1970s reveals that these failures are largely attributed to the poor empirical foundations of EJ analyses, which preclude any authoritative statement on inequitable relationships between racial or income groups, and between environmental problems and associated health burdens. Note, however, that, even where evidence clearly pointed to discrimination, court cases were often unsuccessful as intentional discrimination on the part of the responsible authority or developer could not be proven (Taylor, 1999).

Despite such findings, and difficulties of definition, assessment methodology, and interpretation, EJ is now an important part of environmental and public health policy assessment in the USA, mandated by a presidential executive order (President, 1994). The order requires federal agencies to address EJ as part of their overall mission, and to identify and address disproportionately high adverse human health or environmental impacts of policies, programmes, and activities on minority and low-income populations. The US Environmental Protection Agency, for example, now addresses EJ in planning and decisionmaking, defining 'fair treatment', as that where no group of people bears a disproportionate share of the environmental and adverse health impact of development (Wilkinson, 1998).

Environmental justice in the United Kingdom

In neither the United Kingdom nor Europe more widely is there an EJ movement to compare with that of the USA. However, new European Community laws on enabling rights will ensure that EJ issues are taken more seriously than before. These laws are being driven by the 1998 Aarhus convention (UNECE, 1999), a pan-European treaty that aims to give substantive rights to all EU citizens on three principal environmental matters. Directives on two of these—public access to environmental information, and public participation in environmental decisionmaking—are well advanced in the EU legislative process. The third concern—access to justice in environmental matters—has the objective of giving the public access to judicial and independent procedures to challenge acts or omissions by public authorities and private persons which contravene environmental laws. This area is under formal discussion to clarify the legal standing of groups who might wish to bring a challenge (UNECE, 2002).

In the United Kingdom, which is a signatory to the Aarhus convention, environmental issues which were once firmly allied to a 'green' agenda are increasingly addressed in the political process via concerns for equity and justice. For example, Michael Meacher, Minister for the Environment stated: "Environmental problems are serious and impact most heavily on the most vulnerable members of society, the old, the very young and the poor" (introduction to Boardman et al, 1999) and Jack McConnell, Scotland's first Minister, stated that "people who suffer most from a poor environment are those least able to fight back", and, "I am clear that the gap between the haves and have-nots is not just an economic issue. For quality of life, closing the gap demands environmental justice too. That is why I said ... that environment and social justice would be the themes driving our policies and priorities" (McConnell, 2002). Similar statements have been made by the Prime Minister, Deputy Prime Minister, the leader of the Liberal Democratic Party, and the Chairman of the Environment Agency (quoted by Stephens et al, 2001; SDC, 2002). The link between poor air quality and poverty is a problem recognised by the government's Social Exclusion Unit (Cabinet Office, 2002).

Such statements suggest that EJ interests in the United Kingdom have different social and environmental foci from those of the USA. For example, although the first British EJ-based organisation, the Black Environment Network, did grow from concerns with racial injustice, the principal social justice concern in the United Kingdom relates to environmental inequality and poverty, not race, as was the case in the USA.

As Walker (1998, page 359) argues, this may be because, "given the typical patterns of urban concentrations of ethnic communities in the UK, it would seem unlikely that we have produced a particular racial bias in the siting of hazardous facilities (although this statement remains to be disproved)". Although racial inequalities have been found (Walker et al, 2000), the extent to which they are the result of a racial bias, rather than occurring via an association between poverty and ethnicity remains unclear (Stephens et al, 2001).

Similarly, the environmental dimension of UK EJ research is not focused simply on the location and effects of industrial facilities. After a recent EJ seminar, Stephens et al (2001) concluded that UK EJ research needs to address access to a broad range of environmental resources, including physical needs (shelter, warmth, food, clean air and water); economic needs (transport infrastructure, access to work and services); and aesthetic, mental and spiritual needs (such as quiet or access to the countryside). The government's inquiry into inequalities and health (Acheson, 1998) summarised research that investigated the equity issues of many of these concerns.

The attention to a broad range of environmental issues stems from the view that an EJ framework can be used to develop effective policies and plans via an integrative approach in which social exclusion issues are examined through an environmental lens, and vice versa, by analysing environmental issues from a social justice perspective. Knowing that the majority of road-traffic accidents involving children occur in poor communities (Abdalla et al, 1997), for example, supports measures to reduce vehicle speeds in poor communities, and delivers greater overall social and environmental benefits, including provision of safer play areas, and less accidents, noise, and emissions. Considering environmental issues within a justice framework also facilitates the identification of measures which may bring social and environmental goals into conflict. For example, measures to conserve environmental resources (a domestic fuel tax or compulsory water metering) have both been criticised on EJ grounds because of the added hardship caused to low-income households.

Environmental justice and pollution

Although UK equity research is giving increasing attention to the broader range of issues described above, explicit EJ research continues to be pollution oriented, addressing similar issues to those investigated in the USA: industrial facility location and air pollution. For example, the Friends of the Earth Pollution Injustice campaign found that 662 of the largest factories in the United Kingdom are located in areas with an annual average household income of less than £15 000, with only 6 factories in areas where average annual incomes were greater than £30 000—a very different distribution from that which would be expected if factories were randomly distributed (FoE, 2000). They also found that 82% of carcinogen emissions from Part A processes (large facilities regulated by the Environment Agency) occurred in the most deprived 20% of wards (FoE, 2001). Epidemiological studies in Europe have also suggested a link between elevated rates of birth defects and proximity to landfill sites (Dolk et al, 1998; Elliott et al, 2001). However, although social factors such as deprivation were addressed (deprivation rates were higher for populations within 2 km of a landfill—Elliott et al, 2001, page 20), the social factors were included in these studies because of their role as possible confounding factors, rather than to investigate environmental justice effects.

The greater availability of spatially resolved air-quality data from recent national and local government NAQS modelling studies has allowed analyses of the relationship between demographic indicators and air quality. Table 1 (see over) summarises the main findings of these studies. Only Brainard et al (2002) investigated the relationship

Table 1. UK air quality social equity studies.

Socioeconomic indicator	Location	Observed association with socioeconomic indicator	Reference
Poverty			
Income; car ownership	wards in Greater London	Positive association between deprivation and NO_2 and respiratory diseases.	Stevenson et al (1998; 1999)
Social class index	local authority districts	Weak positive association with PM_{10} and SO_2; very weak positive association with NO_2. Negative association with NO_2 and SO_2 when population density accounted for.	McLeod et al (2000)
Index of multiple deprivation	wards in five cities	Weak positive association with NO_2 and PM_{10} in three cities, inverse in two	King and Stedman (2000)
Index of multiple deprivation	wards in Bradford	Mapped data suggest that NO_2 and PM_{10} "tends to be highest in the most deprived areas".	Pennycook et al (2001)
Index of multiple deprivation	wards in London, Birmingham, Belfast, and Cardiff	Weak positive association with NO_2 and PM_{10} in all cities except Cardiff.	Pye et al (2001)
Various indexes	enumeration districts in Birmingham	Strong positive relationship with poverty, but difficult to separate effect from ethnicity.	Brainard et al (2002)
Social class	West Glamorgan, Wales	No association with NO_2, but analysis of small sample (171 adults).	Lyons et al (2002)
Townsend index	Leeds (3600 point observations)	Strong positive correlation with NO_2.	Mitchell (2002)
Ethnicity			
Percentage of household heads from India and New Commonwealth	local authority districts	Positive association with NO_2, SO_2 and PM_{10}, not attributed to multicollinearity with deprivation measure.	McLeod et al (2000)
Percentage of self-reporting as white, Asian, or black	enumeration districts in Birmingham	Strong positive relationship with ethnicity but difficult to separate effect from poverty.	Brainard et al (2002)
Age			
Pensioners ♀ > 60, ♂ > 65 years; <15 years	enumeration districts in Birmingham	No association with NO_2 or CO emission for any age group.	Brainard et al (2002)

Note: NO_2—nitrogen dioxide, PM_{10}—fine particulates, SO_2—sulphur dioxide, CO—carbon monoxide.

of air quality with age, and found no relationship in the city of Birmingham. They also found that ethnic minority groups were exposed to the poorest air quality, but they could not exclude the possibility of a multicollinearity effect with deprivation. However, working at the local authority district scale, McLeod et al (2000) report a positive relationship between minority ethnic groups and pollution, in which the effect of deprivation is controlled through multilevel modelling.

Most studies investigated the relationship between air quality and deprivation, and tended to show that air pollution is greater in more deprived communities. However, some studies report no association (Pye et al, 2001 for Cardiff; Lyons et al, 2002 in West Glamorgan), whereas McLeod et al (2000) find that, when population density is controlled for, air quality is worst in districts characterised as more affluent. This has led to diametrically opposed conclusions for air-quality management. McLeod et al (2000, page 84) conclude that "measures taken to reduce air pollution in areas of similar population density, for example a city, may actually decrease equity and produce injustice." In contrast, Pye et al (2001, page iv) conclude that "... targeted policies to reduce air pollution concentration in areas where they are high could impact marginally more beneficially in more deprived communities, and therefore move towards reducing the apparent inequity."

Study objectives

The primary objective of the NAQS is to "make sure that everyone can enjoy a level of ambient air quality in public places which poses no significant risk to health or quality of life" (DETR, 2000, page 12). The uncertain and contradictory conclusions of recent UK EJ studies hinders the development of robust environmentally equitable air-quality policies and plans, and highlights the paucity of research in this area. Therefore, the study reported here was conducted to investigate further air-quality – equity relationships in the United Kingdom by testing two commonly held assumptions. First, that those most affected by poor air quality are those least able to do anything about it and, second, that disadvantaged groups bear the costs of pollution which is disproportionately generated by the advantaged.

Data and methodology

Study area and scale of analysis

Because of the release of new data we are able here, for the first time, to study the relationship between air quality and demographic indicators across the whole nation at the ward scale. Our study area is all the wards in Great Britain. Wards are designed to contain roughly equal numbers of electors within local authority districts. Thus ward size is density dependent, with small wards in urban centres and large wards in rural areas. This is a fortunate occurrence as pollutant concentration gradients are generally large in urban centres and small in rural areas. Having small wards in urban centres thus minimises the range of pollutant values coincident with the ward area, and provides a more confident representation of mean air quality for that ward.

In previous studies, national scale analysis has been limited to much larger local authority districts, whereas ward-scale analysis has been limited to individual cities (table 1). Conducting a ward-scale study of all of Britain provides a resolution in which urban scale patterns can be resolved, but in which the potential bias of small sample size incurred by studies of individual cities can be overcome. In total, 10 444 wards were included in the study, covering all of Great Britain: that is, England, Wales, and Scotland, but not Northern Ireland, the Channel Islands, or the Isle of Man, for which not all the necessary demographic data were available.

Pollution exposure

The study focused on nitrogen dioxide (NO_2), a NAQS priority pollutant because of its widespread occurrence at concentrations that give cause for concern on health grounds (DETR, 2000). Health effects of NO_2 exposure include inflammation of the airways, chronic obstructive pulmonary disorders (for example, asthma), and increased reactivity to natural allergens in sensitised individuals. Population-scale impacts are difficult to assess, but indications are that 8700 admissions to hospital for respiratory illness due to NO_2 occur in urban British areas each year (DoH, 1998). Other pollutants, notably ozone and fine particulates (PM_{10}), are also of concern. However, these were not studied as ozone is a secondary pollutant that is not yet well modelled and mapped, and the national distribution of PM_{10} is strongly influenced by transboundary import of pollutants from Europe and wind-blown dust from the Sahara that complicates identification of equity patterns resulting from UK emissions. Other NAQS pollutants (CO, SO_2, benzene, 1-3,butadiene) are generally within permitted levels, or only exceed air-quality standards in very localised areas.

We do note that, although NO_2 presents a major challenge to meeting the NAQS objectives, it is just a single measure of air quality. A full health-impact assessment of air quality would require a consideration of multiple pollutants, as several pollutants occur in the United Kingdom at ambient concentrations that have health impacts and which, collectively, may also have additive effects. In practice, estimation of health impacts both from single pollutants and from pollutant 'cocktails' is constrained by the limited availability of reliable pollutant exposure – health response relationships both for acute and, particularly, for chronic effects (DoH, 1998). Where such relationships are available, health-impact assessment of a single pollutant nevertheless remains difficult, as pollutant exposure is a function of pollutant concentration, and factors of exposure duration, time spent indoors, building characteristics (for example, air conditioning), and level of activity. For these reasons, this study is preliminary in nature, focusing on the concentration of a single, albeit key, pollutant rather than estimated health impacts of all significant pollutants acting in concert. Note, however, that when health responses in populations are calculated, exposure factors are controlled for through selection of exposure – response relationships derived from epidemiological studies of populations, not individuals. In its national assessment of respiratory burden due to NO_2 the government, for example, uses such a relationship, and data on the annual mean pollutant concentration distribution (DoH, 1998). Thus our analysis is also pertinent to equity in health impacts arising from exposure to NO_2.

In correlating annual mean air quality with census data, an assumption is made that an individual's exposure occurs entirely within the relevant ward. Clearly this is a gross assumption and population movement (for example, commuting), introduces a potentially significant bias in pollution exposure—a problem recognised in the air-quality equity literature. The extent of this bias may differ between population groups depending upon their mobility. However, it is thought that the effects of within-day population movement will be less significant when conducting a national scale analysis, as opposed to a local study. Local studies tend to be of major cities within which population movement during the day, through commuting, travel to school, and so on, are greatest.

Nitrogen dioxide mapping

All combustion processes in air produce oxides of nitrogen (NO_x, including nitrogen oxide—NO—and NO_2). In the atmosphere, NO_x are oxidised to form nitrogen dioxide (NO_2), a so-called secondary pollutant. The NAQS objectives for nitrogen oxides are only set for NO_2 (although atmospheric concentrations of NO_x are also measured).

Annual mean NO_2 concentration maps for the United Kingdom in 1999 were provided by the National Environmental Technology Centre. The maps can be viewed in Stedman et al (2001a); Stedman et al (1997; 2001a) describe in detail the methods used to generate them.

The NO_2 maps are based upon emissions recorded in the National Atmospheric Emission Inventory (Goodwin et al, 2000). The inventory provides an estimate of total NO_x emissions in 1998 for a 1 km $\times$ 1 km grid, based upon estimated NO_x emission in over 140 secondary sectors and 9 principal sectors: residential, services, industry, road transport, off-road vehicles, shipping, rail, power generation, and other. Emissions for 1999 were estimated by scaling 1998 emissions at the secondary sector level. Road-traffic emissions, for example, are scaled from projected changes in vehicle activity and emission characteristics under a central growth scenario, assuming current transport policies. Change in vehicle activity (kilometres travelled per year; vehicle speed by road type) is derived from national trip-forecast models, and the change in emission characteristics of the vehicle fleet is a product of changing fleet composition (31 classes defined by vehicle type, age, fuel used) and changing emission factors by vehicle type. Road traffic is estimated to account for 50% of total UK NO_x emission, rising to 75% in urban areas (Goodwin et al, 2000).

Atmospheric concentrations of NO_x are calculated from NO_x emissions by application of a dispersion-box model. The model applies a dispersion coefficient derived from regression of emissions in the vicinity of monitoring sites against the difference between measured NO_x at the monitoring site, and background NO_x taken from a nearby rural site. Annual mean NO_2 concentrations are then calculated with the aid of nonlinear functions relating atmospheric annual mean NO_x to NO_2 for geographical areas with different atmospheric chemistry: rural areas, urban areas, and areas within 3 km of the centre of London. The data upon which these functions are based were collected during 1990–99 using the national automated monitoring network. Verification of the modelled concentrations carried out with the aid of an independent set of measured data collected for 1996–99 shows generally good agreement between observed and estimated annual mean NO_2 (Stedman and Handley, 2001).

Each ward was allocated a value from the NO_2-concentration map as follows. First, ward centroid coordinates were determined for all wards in Britain using a geographic information system (GIS). Ward centroid coordinates were imported to a spreadsheet and rounded (up or down as appropriate) to the nearest 500 m, to match the spatial format of the NO_2 data, the mid-point of each kilometre grid square in Britain. By importing ward centroid coordinates, a corresponding ward identifier, and NO_2 coordinate and value data to relational database, ward centroids were attached to the nearest concentration data point. In a few cases wards shared the same concentration point: this occurred where wards were very small, and the relevant concentration point was the closest to more than one ward centroid. This was considered acceptable as small adjacent wards could reasonably be expected to share similar air quality. No area weighting was used; hence allocation of a ward concentration value from the NO_2 map is more uncertain for the larger rural wards. Seven of the 10 444 ward centroids did not have a NO_2 value within 500 m because of missing data, and so the nearest NO_2 value was allocated manually. For four wards (in Oban, Weymouth, Shetland, and Lowestoft) NO_2 values were allocated from sites 2 km away, and for three wards (in Orkney and the Scilly Islands) NO_2 values were allocated from sites 2–5 km away.

Demographic data

Age

The distribution of pollutant concentration amongst different age groups was examined for two reasons. First, children are largely powerless to influence residential location decisions, and so have, in comparison with adults, a negligible ability to control their exposure to pollutants. Second, children are a priori assumed to be inherently more susceptible to air pollution as their lung function and immunological systems are still developing. We note, however, that, although studies have found greater pulmonary responses to NO_2 amongst infants and children, the available evidence is not sufficiently robust collectively for the government's committee of medical experts on air pollution to recommend NO_2 exposure – response relationships for separate age cohorts (DoH, 1998). Our data on the age distribution of the population came from the 1991 Census, updated so that ward age/sex profiles summed to the published local authority mid-year population estimates for 1999.

Poverty

There are many poverty measures that we could have used in this study (Philo, 1995). However, one measure is particularly suited to this study because it produces an estimate of the numbers of people in each ward who live in poverty, rather than a more abstract index of poverty (Dorling, 1999). This is the Breadline Britain Index (BBI), calculated from statistics from the 1991 Census of Population to estimate the numbers of households living in poverty in each ward in Britain. The estimate was made by Gordon through undertaking a logistic regression of census variables to determine the best combination to use in order to replicate as closely as possible the results of the 1990 detailed Breadline Britain survey (Gordon and Pantazis, 1997). That survey asked its respondents what basic goods people needed in order not to be living in poverty (such as a warm winter coat). Respondents where then classed as living in poverty when they could not own the goods which a majority thought were needed to be living out of poverty. The characteristics of the respondents who were then deemed, on this consensual measure of poverty, to be living in poverty were next used to determine the likelihood of households in the 1991 Census to also be living in poverty. The best estimate of the number of households to be living in poverty for each ward was then calculated as the sum of: 21.7% of households with no car; 20.3% of households not in owner-occupied housing; 16.0% of lone-parent households; 15.9% of households whose head was in social classes IV and V (semiskilled and unskilled); 10.8% of households containing a person with a long-term limiting illness; and 9.4% of households headed by unemployed workers.

In general, this measure of poverty is highly correlated with all other indices of poverty, but correlates more closely with independent measures, such as premature mortality, than any other (even when the illness component is removed). The index is an estimate of the number of households in each area that most people in Britain feel are living below an acceptable poverty line, and thus is extremely pertinent in considering issues of justice. It also has the weakest direct relationship with car ownership of any of the common deprivation measures—a valuable attribute when addressing air quality and vehicle emissions. As 2001 Census data had yet to be released, the index cannot be updated from 1991. However, patterns of poverty tend to change very slowly, and wards which were poor in 1991 are likely to have remained poor in 1999.

Travel and emissions

From their London study, Stevenson et al (1999) concluded that air quality is poorest in areas of low car ownership—a finding used by Friends of the Earth to claim that "traffic pollution is mainly caused by the better off, but the poor feel its effects", and that "traffic pollution is largely caused by richer people living in comparatively clean environments" (Higman, 1999). Mitchell (2002) found a similar relationship for Leeds, but urged caution as, although poorer households do have fewer cars, the cars they own may be older and more polluting. Conversely, ownership of older cars is not limited to the poor (affluent households may own an older second or third car), and affluent people may drive further. Such factors belie the simple interpretation of Friends of the Earth and require further investigation.

We used 1991 Census data to estimate the number of cars to which households had access in each ward in Britain, and the number of households with access to no car, one, two, and three or more cars. We also used data from the Experian data set deposited at MIMAS (Manchester University), which contains a 100% survey of the number of cars registered with the Driver Vehicle Licensing Agency (DVLA) in 1999 by car age and engine size. These data were aggregated from postal sectors to parliamentary constituencies to smooth out anomalies and used to provide profiles of the car fleet identified in the 1991 Census. The DVLA data indicate that the 1991 data were a fair representation of the spatial distribution of cars in Britain almost ten years on.

Annual emission of NO_x from cars within each ward was calculated by grouping the DVLA data into 35 vehicle groups defined by vehicle age (0–1, 1–2, 2–3, 3–5, 5–8, 8–10, >10 years), engine size (<1.4 litres, 1.4–2.0 litres, >2 litres), and fuel type (petrol or diesel). We assumed engine size to be independent of vehicle age. UK vehicle fleet data from MEET (Methodology for Estimating Emissions from Transport) (EC, 1999), the principal EU transport-emission study, were used to estimate numbers of vehicles by fuel type. Heavy-duty vehicles (HDV) are rarely operated by households and were excluded. Light-duty vehicles and motorcycles, which contribute <1% of UK NO_x emissions from transport, were not addressed because of the limited DVLA data, thus the analysis addresses only cars—which constitute 87% of the non-HDV fleet in the United Kingdom.

Each of the 35 vehicle groups were assigned a NO_x emission factor (g km^{-1}) derived from speed-dependent measured emissions over multiple observed driving cycles (EC, 1999). We estimated emission at 55 km h^{-1}, based on MEET UK representative speeds of 25 km h^{-1}, 75 km h^{-1}, and 115 km h^{-1} for urban, rural, and motorway roads, respectively, and corresponding shares of total vehicle kilometres travelled of 46%, 40%, and 14%. Unit emissions are greater for older cars. For example, a 1990 (pre Euro I emission legislation) 1.4 litre petrol engine emits 1.89 g km^{-1} of NO_x; compare this with a similar 1996 Euro I vehicle which emits just 0.14 g km^{-1} (we ignore cold-start emissions). Differences are even greater for newer vehicles, with one pre Euro legislation vehicle emitting the same mass as 30 comparable new cars sold in 2001.

Note, however, that older vehicles are driven less. MEET observations for the United Kingdom show that a new vehicle is driven on average 22 800 km in its first year (1990 figures); this falls to about half this for a ten-year-old vehicle. A Weibull function fitted to the MEET UK distance–age data allowed vehicle age dependent emissions to be corrected for distance travelled each year. Total NO_x emission was then calculated as the sum of emission in each vehicle group, and ward-based emission estimates derived by apportioning these totals according to the numbers of cars by household in each ward within the parliamentary constituency. We did not attempt to use data on travel to work mode, as we are concerned with the residential location of potential polluters and assume that the air-quality data described above account for the influence of travel patterns.

We also note that we have estimated only the NO_x emission from households associated with their use of cars, and have not apportioned other NO_x emission sources, including those from buses, HDVs, and stationary sources, to households. Thus the equity analysis addresses only the relationship between household demographic characteristics and NO_x emission from cars, and not emissions from all transport or all sources. This is considered acceptable, as the purpose of the exercise is to test the assertion (Higman, 1999) described above, that states that it is mainly affluent drivers who pollute poor communities.

Analytical methods

As Wilkinson (1998) and Bowen (2002) observe, there are no standard empirical methodologies for investigating EJ issues. Techniques have included visual comparison of mapped data (Pennycook et al, 2001; Stevenson et al, 1998; 1999), and simple statistical tests [bivariate correlation, t-test, χ^2 (see examples in Bowen, 2002)], but the most common method is regression, including multivariate (Brainard et al, 2002; Jerrett et al, 2001) and multilevel (McLeod et al, 2000) modelling. Bowen (2002) characterises many of the US EJ studies as being of poor or medium quality because of a lack of empirical rigour, allowing no firm conclusions to be drawn. For example, highly significant regression equations are often cited, but with exceedingly low r^2-values. This is not itself a reason to reject a model, but is indicative that there is much unexplained variance, and that tests should be conducted for omitted variables and incorrect functional forms. Bowen (2002) cites Kriesel et al (1996) who showed that race was a significant variable in explaining environmental risk in Georgia and Ohio when race and poverty were the only variables entered into the model, but not when a broader range of variables, relating to education, transportation, and industrial location, were included.

We chose not to conduct regression modelling, given the requirement for data on a very wide range of possible explanatory variables, at ward level for the nation. Rather, we chose to conduct a preliminary investigation of potential inequity by plotting appropriate demographic data in quantiles (with 95% confidence intervals) of equal number of wards (unless otherwise indicated, where equal population was possible). The extremely large number of observations we had to compare (>10 000) meant that this was a relatively robust, simple, method. We also investigated the interaction effects between the distribution of air quality and emissions by examining characteristics of the population resident in wards with differing quantile levels of both rates simultaneously.

Results

Age analysis

The distribution of NO_2 by age in Britain was investigated first. To begin, 256 wards with a zero population were removed (very small part-postcode sectors in Scotland maintained in the census databases for administrative convenience, and wards within the city of London where there are no nighttime residents). For each ward, the percentage of the population in each of thirteen age groups was calculated: <1 year (babies), 1–4 years, then in five-year age bands until 25, and ten-year bands until 79–84 years, and $\geqslant 85$ years. Results were sorted into ascending order for each age group, and placed into deciles, so that the upper deciles are characterised by the greatest proportion of people of the specified age group. The NO_2 values for each ward were then grouped according to the age data and a mean NO_2 concentration for each decile of each age group calculated.

The results (figure 1) show that there are significant variations in NO_2 concentrations within and between age groups. For example, for babies, the upper decile of wards (containing a tenth of all wards sorted to have the most babies), has 18.8%

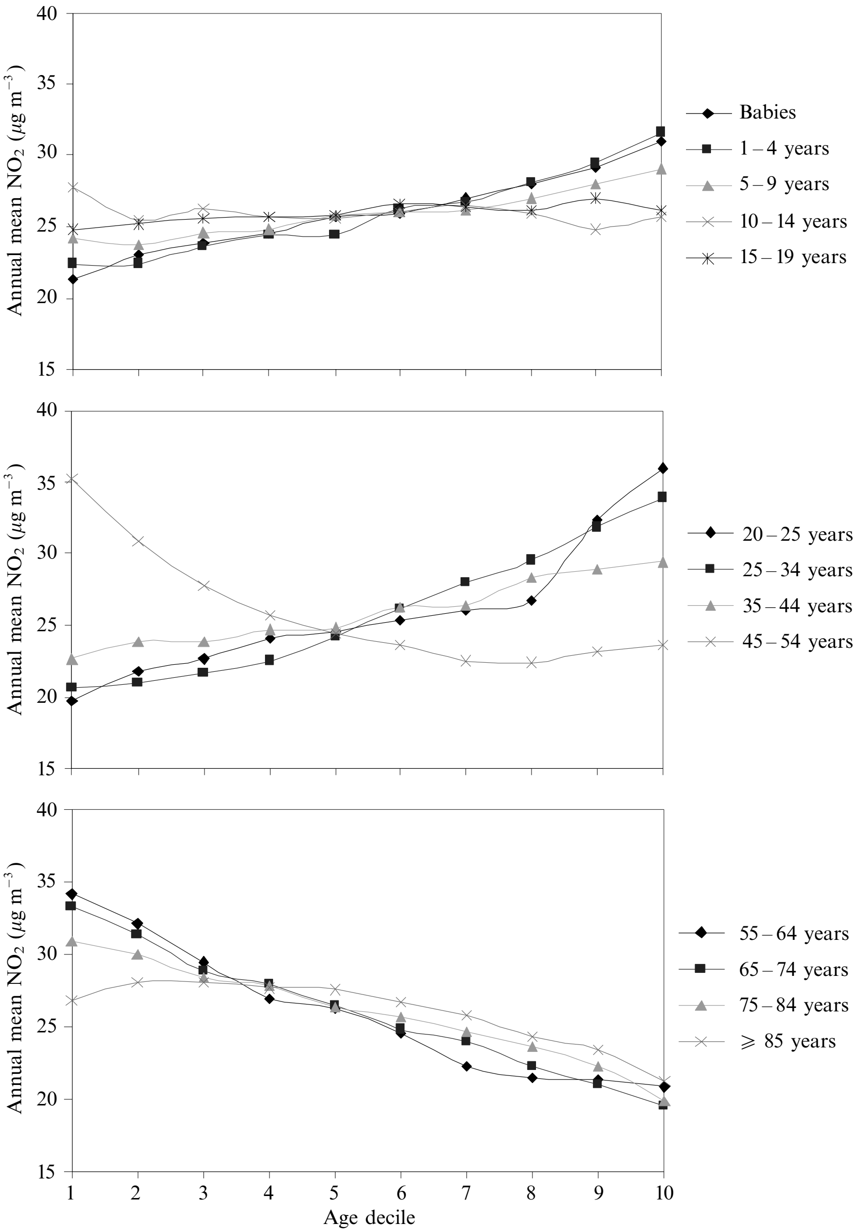

Figure 1. Mean annual NO_2 by age decile for specified age groups in Britain. Age data are sorted in ascending percentage of people of specified age. Each decile has an equal number of wards (10 188). Population and NO_2 data are both for 1999.

of all babies in 1999, and has a mean annual NO_2 concentration of 31 $\mu g\ m^{-3}$. The lower decile, with just 2.5% of all babies, has an NO_2 concentration of 21 $\mu g\ m^{-3}$. Thus wards with a high proportion of all babies also have a much higher mean annual NO_2 concentration compared with the 'few babies' wards. In other words, babies tend to be resident in the most polluted wards. The same is true for children aged 1 – 9 years and for adults aged 20 – 34. Adults aged over 45 were much less likely to be living in highly polluted wards.

From the data shown in figure 1, a ratio of NO_2 concentration in the upper age quantile to NO_2 concentration in the lower quantile was calculated for each age group. Plotting this ratio against age (figure 2) clearly shows how equity in residential exposure to NO_2 varies by age. Wards with a high proportion of babies and infants have approximately 40% higher NO_2 levels than those with few people of this age. This inequity declines with age, with little difference in exposure for teenagers, but wards with a high proportion of 20 – 34-year olds have NO_2 concentrations 60% – 80% greater than wards with few 20 – 34-year olds. The ratio then declines sharply so that wards with a high proportion of adults over 45 years have NO_2 concentrations 20% – 40% less than wards with few over 45s. These results, discussed further below, are readily interpreted in the context of established patterns of migration between urban and rural areas, urban areas being characterised by poorer air quality.

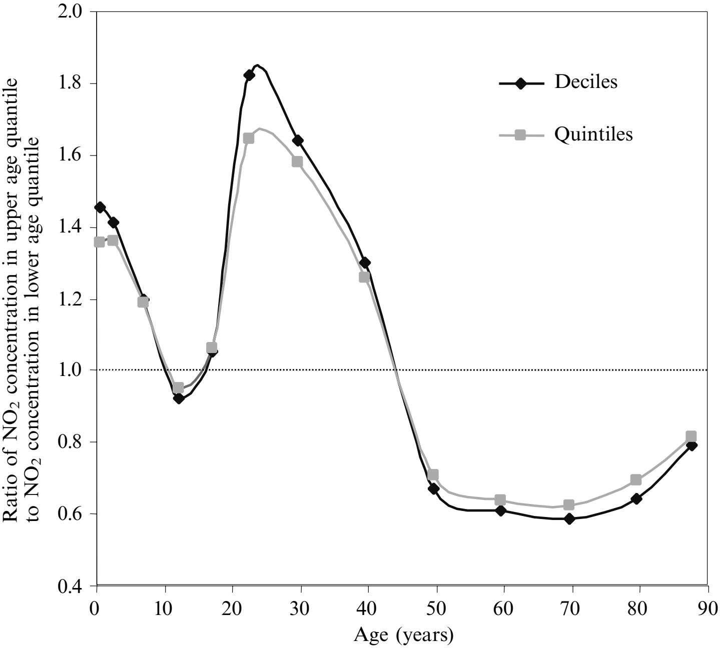

Figure 2. Equity of residential NO_2 in Britain in 1999 showing variation by age. Ratios >1 indicate above-average NO_2 exposure for the specified age.

Poverty analysis

The distribution of NO_2 relative to that of households in poverty was investigated by calculating mean NO_2 concentrations for equal count ($N = 1027$) deprivation deciles (that is, the upper decile represents the 10% most deprived wards in Britain). Figure 3 shows that for the upper seven deciles there is a familiar pattern: a clear positive linear relationship between deprivation and pollution, with the 10% most deprived ward in Britain having a mean annual NO_2 concentration of 32.1 $\mu g\,m^{-3}$, 17% above the national mean. However, the lower deciles also display above-average NO_2, with the 10% least poor wards in Britain having a mean NO_2 concentration of 27.8 $\mu g\,m^{-3}$, 7% above the national average. The distributional implications of this are considered further below. For now, the important point to note is that the relationship between poverty and NO_2 is not a simple linear relationship. The poorest tend to experience the worst air quality, but the least poor do not fair best.

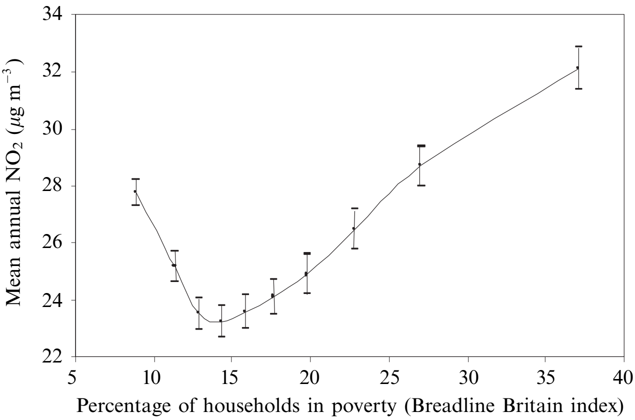

Figure 3. Annual mean NO_2 concentration against deprivation for British wards in 1999. There are 1027 wards per deprivation decile. Bars denote 95% confidence interval (data are normally distributed).

Car-ownership analysis

Many popular deprivation indices, including the Breadline Britain Index used above, include car ownership as a constituent measure of deprivation. However, as road traffic is now the dominant source of UK NO_x emissions (Goodwin et al, 2000), an EJ analysis that looks specifically at NO_2 pollution and car ownership is warranted. Figure 4 illustrates the relationship between NO_2 and car ownership. Wards are again grouped into deciles, each containing 10% of the national population, sorted by increasing car ownership. In the upper decile, where 64.0% of households have no car, mean annual NO_2 is 35.4 $\mu g\ m^{-3}$. In contrast, the lower decile, where only 10.8% of households have no car, the NO_2 concentration is just 22.4 $\mu g\ m^{-3}$. A clear positive linear relationship is evident, with the most polluted wards characterised by low rates of car ownership.

By including people of all ages this analysis is potentially biased, as wards with larger populations of children might reasonably be expected to have lower rates of car ownership, and it has already been demonstrated that wards with above-average numbers of children also have above-average NO_2 concentrations. We therefore repeated the analysis after excluding all children, such that each decile contains 10%

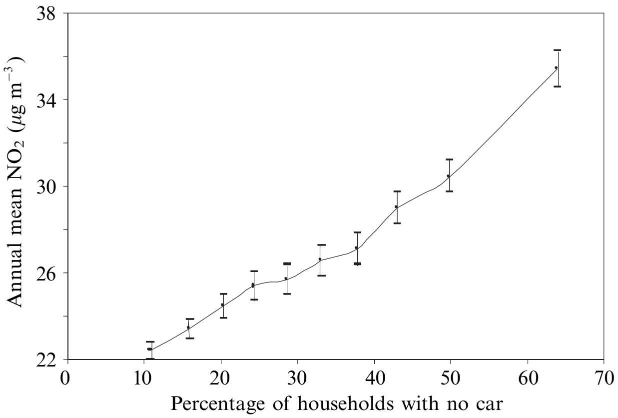

Figure 4. Annual mean NO_2 concentration against car ownership for British wards in 1999. There are 5.78 million people per no-car household decile. Bars denote 95% confidence interval (data are normally distributed).

of all potential drivers ($\geqslant$ 17-year olds), sorted in ascending order of car ownership. The results (figure 5) are very similar to those observed for the entire population, and show that adults in households without a car are more likely to be resident in wards with high NO_2 concentrations than are adults in households with cars. Thus nondrivers are likely to experience higher residential NO_2 exposure than drivers.

The car ownership – NO_2 relationship was investigated further by examining high rates of car ownership by household. From 1991 Census data the percentage of households with three or more cars was calculated for each ward, sorted in ascending order, and placed into equal-count deciles (figure 6). Where there are fewest $\geqslant$ 3-car households (0.8% of households in lower decile) mean annual NO_2 is 26% above the national average, compared with wards with many $\geqslant$ 3-car households (12.5% in upper decile) that have a NO_2 concentration 5% below the national average. Thus households where multiple car ownership is more common tend to be located in less polluted wards, although the upper decile exhibits elevated NO_2 concentration in a similar fashion to the poverty curve (figure 3). This pattern is also seen in the relationship between mean number of cars per household and NO_2 (figure 7).

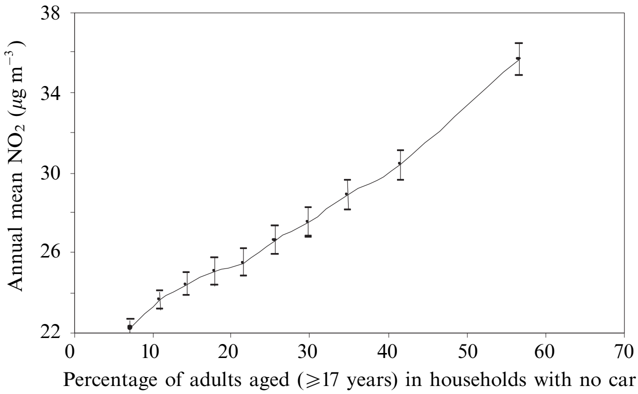

Figure 5. Annual mean NO_2 concentration against car ownership amongst potential drivers for British wards in 1999. There are 4.23 million people per no-car decile. Bars denote 95% confidence interval (data are normally distributed).

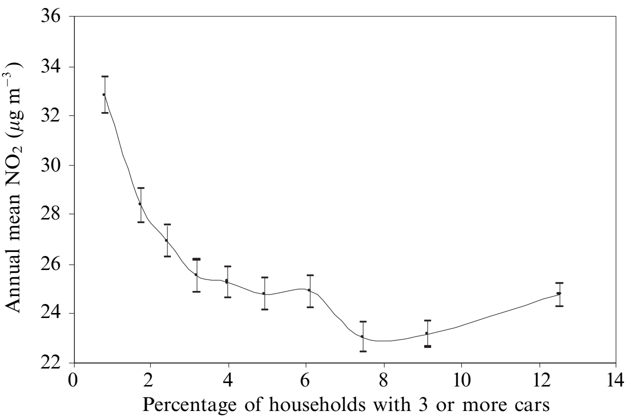

Figure 6. Annual mean NO_2 concentration amongst households with 3 or more cars for British wards in 1999. There are 1027 wards per car-ownership decile. Bars denote 95% confidence interval (data are normally distributed).

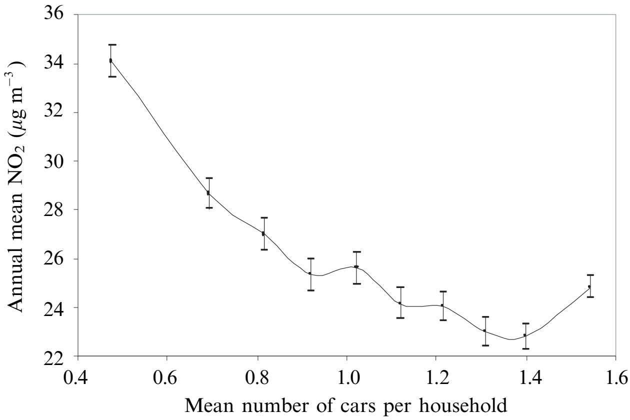

Figure 7. Annual mean NO_2 concentration against mean household car ownership for British wards in 1999. There are 1027 wards per car-ownership decile. Bars denote 95% confidence interval (data are normally distributed).

These relationships warrant further analysis. Why should areas that contain the fewest cars suffer the most pollution? The obvious answer is that they receive a great deal of their pollution from those cars which are driven through or near them by people commuting from areas of high car ownership. Furthermore, population densities are highest and car parking most limited within urban centres where pollution is highest and the need to have a car is also lowest (as urban areas are generally best served by public transport, employment opportunities, and essential services). Thus the relationship between car ownership and area pollution is linear and far stronger and simpler than that with poverty. However, to be more sure that this is what is occurring we need to examine the pollution potential of the cars that people are driving. Not all cars pollute to the same extent. Could it be that the smaller numbers of cars in highly polluted areas are, in fact, highly polluting cars?

The polluter pays?

To investigate the role of vehicle emission in the air-quality–poverty relationship, the ward-level estimates of vehicle NO_x emission were grouped by decile and plotted against their corresponding poverty (figure 8). Although the most polluting wards are most affluent, there is little overall relationship between poverty and emission. Poor and affluent areas could have a similar polluting potential as poor wards have fewer

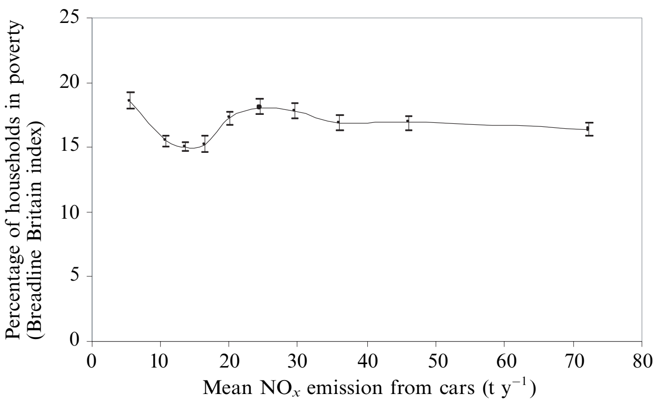

Figure 8. Car NO_x emission against deprivation in British wards in 1999. There are 1027 wards in each decile. Bars denote median and 95 percentile values.

cars that are driven less, but which are older and hence more polluting per se than those of the affluent. Thus it might be concluded that a 'polluter pays' situation operates, with people in areas of poorest air quality contributing most emissions per car. Although it is true that NO_2 concentrations increase with NO_x emission, this is a simplification. By plotting both emission and air quality against poverty simultaneously (figure 9), a pattern emerges in which those wards that emit the least NO_x, but which experience the greatest NO_2 concentrations, are very clearly the most deprived. This indicates that a strong inequality does occur with respect to NO_2 in Britain.

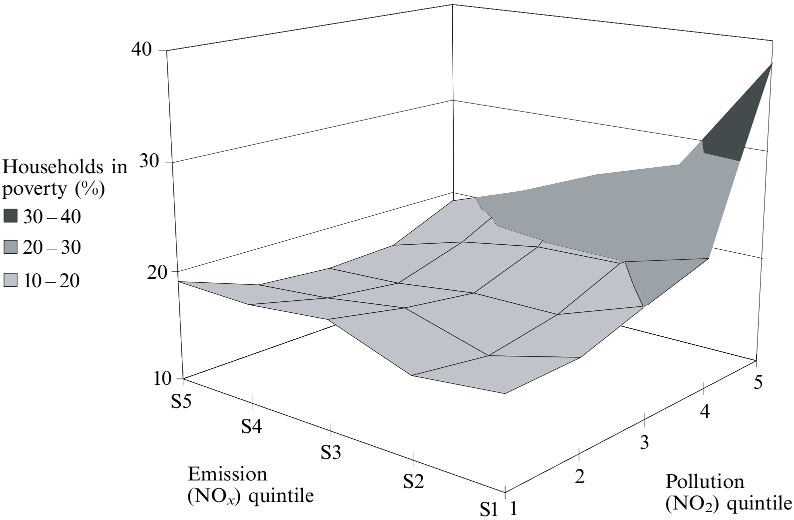

Figure 9. Poverty rate by NO_x emission and ambient air quality for 10 444 British wards in 1999.

Discussion

Our analysis is the first national EJ study to address air quality across the many thousands of local communities in which people in Britain live. It suffers from several drawbacks which we have detailed above, of which the most significant should be stressed before summarising our main findings. First, we are using estimates of air quality and crude estimates of vehicle emissions, which vary spatially and do not account for all ambient NO_2. Second, we have to assume that people's exposure to NO_2 can be approximated by residential NO_2 levels, despite the fact that people often spend much time away from home (although it is reasonable to claim that the very young and very old spend a larger proportion of time at their home, lending credibility to the apparent exposure disparities seen between these age groups in figure 2). Third, we have assumed that the spatial deviation between emission and pollution is largely accounted for by vehicle movement, which would be very complex to address in a national-scale small area study. In short, we have not accounted for the movement either of people during the day or of vehicles, other than assuming that the NO_2 data we have used incorporate the effects of traffic flow (which it is designed to do).

One of our main findings is that different groups of the population are exposed to widely differing levels of NO_2. Most clearly, as people age they are, in general, first exposed to relatively high levels, as more babies are born nearer to city centres than the population as a whole tend to live. As children age, and they and their parents tend to migrate away from city centres, average exposure levels drop, only to rise again as the young adults (that these children become) return towards the city centres when they first find work and/or attend university. From early midlife onwards exposure levels tend to drop, reaching their lowest levels amongst the elderly who are most likely to live furthest away from the centres of pollution. In terms of possible impacts on health this is an unfortunate spatial coincidence as young children and, say, pregnant women (given their age distribution) experience greater exposure than does the population as a whole, although the very elderly, who may also be particularly susceptible to the effects of such pollution, do benefit from their more isolated geographical locations.

We do find that poorer areas suffer higher levels of pollution, as prior UK studies have found (table 1). However, this relationship is the result of a direct and superficially simple inverse linear relationship between NO_2 and the number of cars in each locality. The fewer cars, the higher the pollution, and wealthy areas with few cars (as found within London) suffer just as much NO_2 as poor areas with few cars. However, we also find that for people living in what could be argued to be the most unjust 'polluter-pays' situation, there is a strong link with poverty. People living in wards with some of the highest levels of NO_2, but who contribute little to that pollution, are far more likely to be poor than are the population as a whole (figure 9). Thus for some of the poorest people in Britain there is evidence of environmental inequality which afflicts them more than people of average or affluent means.

We therefore suggest that injustice occurs in two ways. First, there are a set of local neighbourhoods in Britain where people suffer high pollution but contribute little to it. Second, a subset of these people tend also to be much poorer than average. Thus, they suffer the injustice of having to breathe in other (often richer) peoples' exhaust fumes and, being economically disadvantaged, can do little about it (that is, move home). It is an interesting, although largely academic, question to ask what the implications would be had we found that this group of the population tended to be more affluent than the average. Having to live with other peoples' pollution could be argued to be no injustice given their financial freedom to live where they wish (and so trade off air-quality costs against benefits of the locality). This, though, is not the case and it is fair to conclude that the evident inequalities are unjust.

Conclusions

Over forty years ago, Rachel Carson, author of the seminal *Silent Spring* (1962), argued that people should be protected from poisons applied by others into the environment, and that they should have a legal right to redress when that right is violated. Now that the United Kingdom is a signatory to the Aarhus convention on environmental rights, the policymakers of the United Kingdom are obliged to give greater attention to EJ. However, past analyses, with their tentative and conflicting conclusions, have impeded the development of robust air-quality policy mindful of EJ—a problem we have sought to address in this paper.

We have found that those with the least ability to move away from poor air quality (children and the poor) do indeed suffer the greatest exposure. Childhood exposure is a product of parental location choices and, although there is a clear age-related inequality, it is debatable to what extent this is unjust given that parents are presumably making location choices intended to maximise family welfare. More obvious injustice occurs with respect to the poor. However, even here the injustice is not clear-cut, as some affluent households also suffer high exposure. The group that suffers most from the injustice of air pollution, given the distribution above, is the children of the poorest wards in Britain who live in areas of very low car ownership.

The real injustice becomes apparent when one considers whether the 'polluter-pays' principle operates, that is, whether those suffering poor air quality contribute significantly to it. First, it is apparent that pollution exposure rises with falling car ownership. This is a much stronger relationship than that between poverty and pollution, where the most and least affluent both experience above-average exposure, and is likely to explain the weak pollution–deprivation relationships found in the recent government-sponsored study (Pye et al, 2001). However, when car ownership is replaced as an indicator of pollution contribution by the preferred estimate of emission, it is apparent that emission and poverty are much more weakly related, and that in general across Britain the poor contribute a significant proportion of the pollution that they are exposed to (although we do appreciate that drivers who lack the income to move to a cleaner area may also be unable to purchase cleaner vehicles). Thus it cannot be claimed that the poor bear the pollution costs of the rich, as has been stated based on the London study of Stevenson et al (1998). The exception, however, is for a minority of the poor who experience high pollution exposure but who contribute little in the way of emissions. This exceptional group, highlighted in figure 9, appears to be living in a situation that is patently unjust.

Although the results of this analysis may appear intuitive to some, they cannot be taken as a given, and it is probable that other developed countries—those with higher rates of rural poverty, for instance—do not exhibit the patterns that we have found. However, given these findings, the logical next question to ask is that of what should, or indeed could, be done to address such inequalities. Before asking this, it is worth reiterating that a variety of methods have been applied to assess environmental inequalities and that, before addressing the question of policy responses, it is important to reach a broad agreement over acceptable methods and hence over findings. Thus a degree of consensus is needed over treatment of scale and spatial coverage, selection of appropriate demographic and environmental metrics, how to treat cumulative and indirect impacts, and which justice framework to use to interpret any inequalities.

Suitable responses to inequalities and injustice may be difficult to identify, particularly given that the basic urban and social geography of Britain changes slowly, despite processes such as brownfield development and limited inner-city gentrification. However, if plans and policies are to be assessed with respect to sustainability objectives, then the environment–society trade-off cannot simply be ignored in assessments because it is difficult to remedy inequalities. A first response must be to ensure the reasons for any inequalities are adequately understood. For example, are air quality–poverty relationships a result of declining air quality in poor communities, or of a neighbourhood transition where the more affluent move away from a polluted area? A 'do nothing' policy response to an inequality requires such an understanding if the motivation for the response is to be accepted by the affected community. Processes which maintain or exacerbate an existing regressive situation also merit attention (Walker, 1998). Blowers (1993), for example, warns of the creation of 'hazard havens' where new hazards are located close to existing ones on the grounds that they may be less prone to NIMBY ('not in my backyard') responses.

We note that UK air quality is predicted to improve further over the next decade or so, and that many fewer people will be subject to exceedences of air-quality standards. However, the spatial distribution of pollution will remain much as it is now (Stedman et al, 2001b), and hence inequity patterns are also likely to remain largely unchanged. Given that air-quality standards are set with reference to economic objectives, with health impacts arising at *all* concentrations (DoH, 1998), injustice apparent today will not be remedied by a 'do nothing' policy. Furthermore, as time passes, more detrimental effects of pollutants at lower concentrations tend to be found. However, numerous policy and planning responses to improve air quality are detailed in the NAQS (DETR, 2000). The environmental equity effects of one key measure, road-user charging, has been investigated for one city (Mitchell, 2002) but the environmental equity implications of most of these measures are largely unknown. Indeed, the appropriate focus for such assessments is open to debate, as evidenced by our findings above (figure 9) which suggest that the focus of air-quality policy directed at tackling injustice should have a polluter-pays dimension, rather than merely focusing on poor air quality in deprived areas, as recommended to government by Pye et al (2001).

Ultimately, the appropriate response to air-quality inequalities can only be determined through a planning and assessment process that makes communities aware of environmental inequalities and provides them with access to the planning system to help address them. The first two pillars of the Aarhus convention, giving rights on access to environmental information and public participation in decisions that affect the environment, are the subject of draft European Union directives. The third pillar, access to justice in environmental matters, is currently subject to consultation to determine who is eligible to make a complaint. The results of that consultation could have extremely wide-ranging implications.

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