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Forecasting Environmental Equity: Air Quality Responses to Road User Charging in Leeds, UK

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Forecasting Environmental Equity: Air Quality Responses to Road User Charging in Leeds, UK

Abstract

Sustainable development requires that the goals of economic development, environmental protection and social justice are considered collectively when formulating development strategies. In the context of planning sustainable transport systems, trade-offs between the economy and the environment, and between the economy and social justice have received considerable attention. In contrast, much less attention has been paid to environmental equity, the trade-off between environmental and social justice goals, a significant omission given the growing attention to environmental justice by policy makers in the EU and elsewhere. In many countries, considerable effort has been made to develop clean transport systems by using, for example, technical, economic and planning instruments. However, little effort has been made to understand the distributive and environmental justice implications of these measures. This paper investigates the relationship between urban air quality (as NO_2) and social deprivation for the city of Leeds, UK. Through application of a series of linked dynamic models of traffic simulation and assignment, vehicle emission, and pollutant dispersion, the environmental equity implications of a series of urban transport strategies, including road user cordon and distance based charging, road network development, and emission control, are assessed. Results indicate a significant degree of environmental inequity exists in Leeds. Analysis of the transport strategies indicates that this inequity will be reduced through natural fleet renewal, and, perhaps contrary to expectations, road user charging is also capable of promoting environmental equity. The environmental equity response is however, sensitive to road pricing scheme design.

Keywords: Air quality, environmental equity, environmental justice, transport planning, road pricing.

1 Introduction

1.1 Environmental equity and justice

Sustainable development has three widely agreed meta-goals: sustained economic development (inter-generational equity), environmental protection, and social justice (intra-generational equity) (WCED, 1987). Because there are trade-offs between these goals, all three must be addressed together if development is to be sustainable. Feitelson (2002) observes that, whilst trade-offs between economic development and the environment, and between economic development and social justice have received considerable attention, much less attention has been paid to the trade-off between environmental and social justice goals. Furthermore, this trade-off, often referred to as environmental justice (EJ), has rarely been coupled with issues related to transport.

As Agyeman and Evans (2004) note, EJ is a contested concept with many possible definitions. A recent definition is that contained in the US Commonwealth of Massachusetts EJ policy, which states that:

> "Environmental justice is based on the principle that all people have a right to be protected from environmental pollution and to live in and enjoy a clean and healthful environment. Environmental justice is the equal protection and meaningful involvement of all people with respect to the development, implementation and enforcement of environmental laws, regulations and policies and the equitable distribution of environmental benefits
>
> Commonwealth of Massachusetts (2002)

Agyeman and Evans (2004) note that this definition implies that EJ has "procedural ('meaningful involvement of all people') and substantive ('right to live in and enjoy a clean and healthful environment') aspects" and that "unlike most definitions, it makes the case that environmental justice policy should not only be reactive to environmental 'bads', but should also be proactive in the distribution and achievement of environmental 'goods' (a higher quality of life, a sustainable community)". Other definitions of EJ are less explicit with respect to the procedural dimension, and emphasise the distribution of environmental quality. Cutter (1995), for example, 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 socio-economic status". No attempt is made here to further define EJ or address EJ directly, but a clear distinction is drawn between EJ and environmental equity, the focus of the paper.

Environmental equity here refers to the social distribution of environmental quality (and specifically the distribution of $N0_2$ by deprivation status). In contrast, EJ must also consider to what extent the observed distributions are 'unfair'. One element of this interpretation is a consideration of how a particular distribution has arisen. Whilst such causality issues are poorly addressed in empirical EJ studies to date, numerous mechanisms by which an unequal distribution may arise have been postulated, ranging from deliberate discrimination within the planning system to natural socio-economic processes relating to neighbourhood change (for example, people may choose to locate in an area of low environmental quality to take advantage of local employment opportunities or a better quality house).

A second element in the consideration of fairness is the justice theory subscribed to by those making the EJ assessment. That is, for a single distribution, different conclusions as to 'fairness' may be made depending upon whether the assessors consider a just distribution to be one where people get what they need, what they have a right to, or what they deserve. Thus understanding causality and the justice theory applied are key elements in the interpretation of environmental injustice. Such considerations (see Capek 1993; Cutter 1995; Liu 2001; Walker and Mitchell 2003 for further discussion), are however, largely beyond the scope of this paper which addresses a more limited, but essential first step in EJ assessment, the identification of the social distribution of environmental quality, here after referred to as environmental equity assessment.

1.2 The Emergence of Environmental Justice

Environmental justice issues have received significant attention at the global level, most notably with respect to the relationship between developed and developing countries. Research in this field has, for example, addressed differential contributions to, and impacts of, climate change, and the distribution of the costs and benefits of natural resource exploitation, both issues where transport is important (Bhaskar, 1995). Local scale environmental equity issues, of the kind addressed by this paper, are in comparison much less studied. However, policy developments at the highest level (e.g. a Presidential order in the USA; a UN ECE convention on the environment) mean that in future, greater cognisance of local and regional environmental equity issues is required when evaluating projects, plans and policies that affect the environment.

In the USA, the analysis of EJ is now an important part of environmental and public health policy assessment. The US Environmental Protection Agency, for example, now addresses EJ in their National Environment Policy Act (NEPA) planning and decision-making process, defining 'fair treatment', as that where no group of people bear a disproportionate share of the environmental and adverse health impact of development (US EPA, 1995). This action was mandated by President Clinton's Executive order 12898 that directed "All Federal agencies to make environmental justice part of their mission, and to identify and address disproportionately high and adverse human health or environmental effects of their programs, policies, and activities on minority populations and low income populations" (President Clinton, 1994). A memorandum accompanying the order also requires that Federal agencies ensure that communities have access to relevant information and are given opportunities to effectively participate in agency actions that affect them.

These EJ responsibilities developed from the concerns expressed by civil rights activists in the 1970's and 1980's, who demonstrated that landfills and polluting industries were disproportionately sited within predominantly black communities or indigenous peoples' reservations (Bullard, 1990; Lavelle and Coyle, 1992). However, class actions brought against civil authorities on the grounds of unjust planning decisions have proved largely unsuccessful, for two reasons. Firstly, poor empirical foundations of EJ analyses have precluded authoritative statements on inequitable relationships between racial or income groups, and environmental problems and associated health burdens (Bowen, 2002). Secondly, where evidence has clearly pointed to environmental inequity, intentional discrimination on the part of the responsible authority or developer has rarely been proven (Taylor, 1999). Although the Presidential order creates no legal rights, litigation will be an important mechanism in determining how environmental inequities are determined and evaluated within the justice framework created by the order.

In Europe and the UK, EJ issues are also attracting significant attention. Recently, EC directives have been passed on access to environmental information (2003/4/EC) and participation in environmental decision making (2003/35/EC). These directives were introduced to meet the provisions of the UN Economic Commission for Europe (UN ECE 1999) "Convention on Access to Information, Public Participation in Decision Making and Access to Justice in Environmental Matters" (the Arhus convention), which came into force in 2001. A third EC directive on access to justice in environmental matters is proposed, and 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 currently under formal discussion to clarify the legal status of groups who might wish to bring a challenge on environmental justice grounds (UN ECE, 2002).

Although the UK does not have an EJ movement to compare with that of the USA, interest in the field has grown rapidly in the last five years. The discourse on EJ has been lead by academics (Dobson 1998; Walker, 1998; Agyeman 2000), NGOs (Boardman *et al*., 1999; SDC 2002; Adebowale 2003) and pressure groups (FoE 2001; Dunion 2003). These activities have supported the strong policy guidance from the EU, leading government to voice strong support for the principle of EJ, although this has not yet been translated into significant activity at the regional and local levels. A review of this emerging discourse is provided by Agyeman and Evans (2004), who conclude that the links between EJ and sustainability are becoming clearer and more widely understood in the UK, both by government and others.

This understanding has been fostered by empirical studies into the relationship between environmental quality and social distributions. Friends of the Earth (FoE, 2001) conducted the first analysis of this type in the UK as part of their 'Pollution and Poverty' campaign, and concluded that the large polluting factories were disproportionately located in poor communities. Many similar studies have followed, including substantive small area national analyses for the Environment Agency in England and Wales (Walker and Mitchell, 2003) and on behalf of a group of Scottish NGO's (Fairburn *et al*., 2004). Whilst the conception of EJ in the UK is broader than that of the US (e.g. it addresses access to environmental 'goods' and fairness in procedural matters), most studies have similarly focussed on environmental pollution, as adequate small area data to support other analysis is generally poorly available. Whilst the evidence base for environmental injustice in the UK remains comparatively weak, a review of past research conducted for government, concludes that "In the UK, environmental injustice is a real and substantive problem…that afflicts many of our most deprived communities and socially excluded groups" (Lucas *et al*., 2004).

1.3 Environmental Equity and Transport

Equity issues addressed in transport have been largely concerned with equity-economy trade-offs, including the relationship between public and private transport, the impact of transport investment on peripheral areas, and the effects of transport investments and policies on specific, underprivileged population groups (Feitelson, 2002). However, transport is also an important determinant on environmental-equity relationships. It produces direct effects such as atmospheric emissions and noise, and also indirect effects, through its influence on the location of polluting facilities and affected people.

Environmental-equity issues have been little studied within a transport context, and to date, point sources have provided the focus for most environmental-equity studies. Cutter (1995) and Bowen (2002) review North American studies, all of which address associations between emissions from industrial facilities and landfills with attributes of nearby populations. Early European studies share this focus, also investigating toxic emissions and landfill sites (Dolk *et al*., 1998; FoE, 2000; FoE 2001; Elliot *et al*, 2001). In the UK however, several EJ air quality studies, in which transport emissions are a key factor, have also been conducted (see Table 1 and a review in Mitchell and Dorling, 2003).

TABLE 1 about here

These studies have all sought to assess the current social distributions of air quality, with little consideration of how these patterns may change in future, or response to policy or plan intervention. Such environmental-equity analysis of alternative transport strategies has been very limited to date. A notable exception is the SPARTACUS project, where a land use-transport interaction model was applied to three European cities, and the impact of different land use and transport policies on exposure of socio-economic groups to transport emissions determined (LT *et al*., 1998). Forkenbrock and Schweitzer (1999) applied models of pollutant emission (EPA MOBILE5 and PART5) and dispersion (CAL3QHCR) to derive air quality maps for a neighbourhood intersected by a main arterial highway. The air quality and noise maps (the latter modelled using MINNOISE) were superimposed on socio-economic data by census block within a GIS, and the proportion of low-income and minority populations in different exposure bands determined. However, this demonstration project made no attempt to assess the equity impacts of alternative transport policies. For Los Angeles, Bae (1997) evaluated the welfare benefits (health, property value, unemployment risk etc.) arising from the realisation of federal clean air standards, and concluded that low income and minority groups would benefit disproportionately, and hence that the air quality policy was progressive.

However, transport was only addressed indirectly, assumed to be a key factor in achieving the desired air quality standards.

Feitelson (2002) argues that EJ research should not follow past equity studies outside the transport arena, which have simply compared affected areas to unaffected areas, and so failed to produce meaningful and robust results able to guide balancing of the three sustainable development meta-goals. Rather research should introduce the equity dimension into evaluations of exposure to environmental impacts arising from alternative transport policies and plans. This recommendation is consistent with the 'New Approach to Transport Appraisal' (NATA) adopted by UK government (DETR, 1998a) and the supporting appraisal guidance (DfT, 2005). The guidance requires an assessment against a series of economic, social and environmental goals, and a number of 'supporting analyses', which includes 'distribution and equity'. This equity analysis is intended to show the distribution of impacts geographically, by transport mode, and by social group. Detailed guidance on how to undertake the distributional analysis is not yet available (although some examples relating to the use of GIS in noise and air quality analysis are given), but the guidance does indicate that environmental equity analysis is an important and developing aspect of UK transport policy and plan appraisal.

To date, all the UK EJ studies (air quality or otherwise) have been cross sectional, identifying current environment-equity relationships. None have a longitudinal component in which the impact of alternative development strategies on environmental equity is assessed, as recommended by Feitelson (2002) and by the NATA guidance. This paper therefore describes the first study in which the environmental equity implications of alternative transport strategies are assessed for a UK city. The strategies investigated are relevant to local government's currently seeking to address problems of urban congestion and pollution.

2 Methods

2.1 Assessing the Air Quality Impacts of Road Transport Strategies

In response to the 1996 EU Air Quality Framework Directive (96/62/EC), the UK has developed a National Air Quality Strategy (NAQS) (DETR, 2000) that defines UK policy, tasks and responsibilities for achieving ambient air quality objectives. The focus on air quality, rather than emissions, requires local government's to assess compliance with air quality standards, and usually requires application of dispersion models. Where exceedence of standards for the 2005 target date are forecast, air quality management areas (AQMA's) must be designated, and an action plan developed detailing measures that the authority intends to pursue in order to achieve the prescribed air quality objectives.

The government's 1998 Transport White Paper (DETR, 1998a) identifies transport planning as a key mechanism by which the NAQS objectives can be met. However, to date, there is limited empirical evidence available to inform local governments of the impact that different transport options may have on urban air quality, particularly at the strategic rather than project level. Thus information on the environmental gains of alternative transport plans is required to support the local authority transport and NAQS planning processes.

To address this demand, air quality in Leeds, UK was assessed under a range of strategic road transport options of interest to local and central government. Leeds is a city of approximately 750,000 people in northern England with a mixed economy that has grown faster than any other city in the country in the last decade, contributing to growing congestion problems. The options investigated were: (1) do nothing, assessed for the years 1993, 2005 and 2015; (2) road user charging under a single inner cordon with a £3 toll; (3) road user charging under a double cordon, with a £1 outer cordon charge and a £2 inner cordon charge, giving the same £3 toll to enter the city centre as the single cordon; (4) road user charging under distance based charges of 2, 10 and 20 p per km travelled within the zone outlined by the outer cordon; (5) network development, including 7 km of urban dual carriageway intended to ease city centre traffic congestion and provide access to a new economic development zone; and (6) promotion of clean fuelled vehicles to 2015.

The air quality assessments were made through application of TEMMS (Transport Emissions Modelling and Mapping Suite) a GIS-Model that integrates component models of traffic simulation and assignment on urban road networks (SATURN) (Van Vliet, 1982); emission from road vehicles (ROADFAC); and atmospheric dispersion (ADMS-Urban) (CERC, 1999) to give pollutant concentration maps. A detailed description of TEMMS, including the ROADFAC model and its integration with SATURN and ADMS-Urban, as well as the calibration and validation of the full system is provided by Namdeo *et al*., (2002). Its application to assess respiratory disease burden response to road transport emissions has previously been described by Namdeo *et al*. (2000) and Mitchell *et al*., (2000).

The application of TEMMS to each of the road transport strategies listed above is described in detail in Mitchell *et al.,* (2002a, 2002b), hence only a brief description is provided here. Except where otherwise stated, all assessments addressed the NAQS target year of 2005. For each scenario, the same stationary source emission inventory was used, detailing over 400 point source emissions. Area source emissions and background imports were also held constant for all tests; hence differences in air quality between scenarios were due solely to transport effects. Vehicle responses to road user charges were assessed through application of SATURN's SATTAX module (Milne and Van Vliet, 1993). A variable trip matrix was used with demand responses applied using an exponential function, calibrated by stated preference survey data, with elasticity values of 0 to -1.0 for most conceivable changes in generalised cost. Thus the SATURN forecasts represented the combined driver responses of re-routing and changes in demand. The SATURN model represented all vehicles as passenger car units (PCU) hence no attempt was made to differentiate between vehicle types (which have different emissions) in the road user charge tests.

Emission factors and UK fleet composition data for future years to 2015 were drawn from the final report of the EU MEET working group on emission estimation (EC, 1999). A total of 72 vehicle classes were represented addressing vehicle type, weight, engine capacity, fuel type and emission control technology. Additional cold start emissions were estimated using CORINAIR methods (Eggleston *et al*, 1991). Morning peak hour emissions were calculated for each of the 10,250 links in the Leeds road network. Link emissions for all other hours in the day were derived by applying a time variant factor to the morning peak emission, derived from observed vehicle count and speed data collected hourly throughout the week for a representative range of road types.

Emissions were modelled for a 30 x 25 km box centred on Leeds, and pollutant concentrations simulated for an inner 12 x 12 km box (at 200 m intervals) covering the entire built area. Sequential (hourly) meteorological data was used as this gives a better estimate of peak concentration values than statistically averaged data. NO_X to NO_2 conversion was calculated using the ADMS-Urban Generic Reaction Set. In hourly steps, the dispersion model simulates one year of pollutant concentrations from which compliance to the prevailing air quality standards, as annual mean and percentile values, is assessed. Results from the clean fuel scenario were found to be highly uncertain due to inadequate emission factors for LPG Euro II-IV vehicles at typical urban speeds; hence these results were not carried forward to the environmental equity analysis described below.

2.2 Environmental Equity Analysis

The analysis reported here is based upon NO_2 concentration as the environmental variable. Nitrogen dioxide (NO_2) was selected as the study pollutant, as NAQS studies have indicated that NO_2 and PM_{10} are the principal pollutants of concern in UK urban areas, and are thought to pose significant risks to health. In addition, NO_2 in Leeds is more sensitive to changes in transport emissions than PM_{10}, due to a large point source contribution to total particulate emission. The NO_2 24-hr annual mean value is used in preference to the percentile value, as this parameter is recommended by COMEAP, the UK committee of medical experts on air pollution, for use in respiratory disease burden estimation (DoH, 1998).

Annual mean NO_2 concentration is reported at each of 3600 points spaced at 200 m intervals in a grid cell pattern throughout the 144 km^2 inner box. The disease burden response (hospital admissions and deaths brought forward due to NO_2) to ambient air quality was quantified in the main study (Mitchell *et al*., 2002b), but was not subject to an equity analysis. This was because the disease burden function recommended by COMEAP, which translates pollutant concentration to disease burden, is a linear, through the origin, no threshold dose-response relationship. Thus an analysis of disease burden would produce the same equity pattern as that simply using air quality data.

Modelled NO_2 concentrations were then related to a poverty measure, so as to identify equity relationships. Most North American environmental equity studies address ethnic minorities, but for this analysis, poverty was the preferred measure, for two reasons. Firstly, public pressure to address environmental injustice against ethnic minorities is largely absent in the UK. This is exemplified by Ministerial statements on EJ in the UK (Mitchell and Dorling, 2003) where concern is expressed for vulnerable people least able to help themselves. These are seen as the very young, the old and particularly the poor. Secondly, demonstrating that environmental inequalities occur with respect to race is complicated due to co-linearity between race and income. Bowen (2002) cites examples from the USA, where evidence for a racial bias in hazardous facility siting was discounted once income related factors were considered. Evidence that ethnic minorities in the UK are exposed to greater environmental hazard is also very limited, although this is admittedly a little researched area. At present, poverty is the principal focus of EJ concern in the UK.

For each grid cell in the Leeds model, demographic data was obtained from the MIMAS database held at Manchester University (Anon, 1999) so as to quantify poverty. The grid cell data is generated from 1991 census enumeration district data using the SURPOP modelling technique (Bracken and Martin, 1989; Goodchild *et al.*, 1993). This technique is designed to reduce the problem of falsely representing continuously variable spatial data when using polygons whose attribute values are spatially uniform. Using SURPOP demographic data meant that the equity analysis could be much more sensitive to localised changes in air quality than if using census districts. However, the SURPOP model has not been applied and tested for all UK census variables, which limited the range of deprivation measures that could be used in the equity analysis, including the index of multiple deprivation now favoured by government.

Poverty was therefore measured using the Townsend Material Deprivation Index (Townsend *et al.*, 1988). The Townsend index comprises the following variables: Unemployment (residents over 16 as a percentage of all economically active residents over 16); Overcrowding (households with one person per room and over as a percentage of all households); Non-car ownership (households with no car as a percentage of all households); Non-home ownership (households not owning their own home as a percentage of all households). The unemployment and overcrowding variables tend to be positively skewed and so are log transformed. All variables are then standardised using Z-scores, where for each cell the score is the sum of the observed cell value less the population mean divided by the population standard deviation. Z-scores are standardised using Leeds data only, and so must not be compared to those for other locations. Positive values represent relative deprivation, negative values relative affluence. The data reveals that inner city Leeds is most deprived, and the suburbs are most affluent, although this is something of a generalisation.

There are several caveats associated with the data. Firstly, annual mean NO_2 concentration can only act as a surrogate for exposure to NO_2, itself a surrogate for air quality more generally, and of health impact. Individuals have different personal exposures depending on their activity rate (which may be linked to demographic variables), time spent indoors, structural characteristics of buildings (e.g. ventilation), and their proximity to pollution sources. Whilst the air quality data used here is modelled with a spatial resolution of 200m intervals, more detailed spatial analysis using the ADMS-Urban options of 'intelligent gridding' and street-canyon analyses demonstrated that NO_2 concentration displays considerable heterogeneity within cells, and that at a micro scale, exposure can vary markedly over a few tens of metres (Mitchell *et al*., 2002b). Also, note that no sensitivity analyses were conducted, due to the very long run times (circa 30 days) to complete dispersion modelling for each scenario.

Secondly, the environmental equity analysis pairs NO_2 concentration data with deprivation data based on residential location. Clearly people do not spend all of their time at home, hence the use of site mean concentration data is further limited as a surrogate for exposure and health impact. This analysis, in common with all other air quality and equity studies to date, does not address the issue of within day movement, due to difficulty in making reliable day and night time population estimates. Data required to improve the exposure estimates, such as travel to work statistics for example, is of limited value due to the coarse scale for which it is available in the UK. Trip matrices from the SATURN model could be used to develop a better understanding of day and night time populations in Leeds, but this was beyond the scope of the analysis. However, such data would not include the socio-economic information required to determine the deprivation status of this travelling population, and hence their inequality in exposure.

The deprivation data places a further caveat on the analysis, in that it is derived from the 1991 census (2001 census data not yet available at the time of writing), whilst the majority of the air quality data is modelled for 2005, the NAQS assessment target date. Thus for much of the analysis, a substantial temporal 'gap' exists between the principal data sets, which could distort the observed equity patterns. However, under the do-nothing scenario, air quality was mapped for 1993 (the earliest year for which a trip matrix was available), sufficiently close to the 1991 census date to give high confidence in the relationship between deprivation and air quality for the base year. Deprivation - air quality relationships for subsequent scenarios are modifications of this basic 1993 relationship, driven by transport induced change in air quality. Finally, as the poverty data is constant for all scenarios, it does not reflect possible changes in urban structure from 1993 to 2015. However, the spatial pattern of deprivation in Leeds is not thought to be changing significantly, and that changing urban structure is anyway only relevant to the do-nothing scenario, as the others all share a common base year.

3 Results

Figure 1 illustrates annual mean NO_2 in Leeds for 2005 under the do-nothing option, whilst Table 2 summarises the spatial re-distribution of NO_2 in response to the modelled transport scenarios. Clearly redistribution of pollution occurs by location, with some sites experiencing an improvement in air quality, and others a decline. However, further analysis is needed to assess the environmental equity associated with the transport plan options. Two statistical tests were used. Firstly, for each scenario, an ordinary least squares regression was conducted of annual mean NO_2 and the deprivation index. The regression is not used to infer causality between variables, but is used to test for association between them, with a steeper slope coefficient indicative of greater sensitivity of Townsend scores to NO_2 levels. The use of regression models where causality is not inferred is discussed by Cook and Weisberg (1999), and has previously been used in an air quality-equity context by Jerrett *et al.*, (2001). Secondly, difference tests are conducted which compare annual mean NO_2 concentration in the upper and lower quartiles of the deprivation index.

TABLE 2 about here

FIGURE 1 about here

The relationship between social deprivation and NO_2 under the do-nothing scenario is shown in Figure 2, and the environmental equity effects of road network development and road user charging in Figures 3 and 4 respectively. For each plot, the data are presented as the mean of cell annual mean NO_2 concentrations (ug/m^3) grouped by Townsend score integer classes. There are few Townsend values < -5 (N=24), so these observations are grouped. There are a total of 1851 observations, not 3600, as there are many cells where there is no resident population. In all cases, a strong positive association between deprivation and NO_2 is apparent. This is most marked in the 1993 case, where NO_2 concentration in the most affluent locations (Townsend score < -5) is 14% below the city average, whilst in the most deprived areas (Townsend score > 5) mean annual NO_2 concentration is 23% greater than the average. This represents a difference of 10.6 ug/m^3 NO_2 between the most deprived and affluent communities in Leeds, significant in the context of an annual mean NO_2 health based standard of 40 ug/m^3, and the COMEAP dose-response relationship of 0.5% increase in hospital admissions for respiratory illness for every 10 ug/m^3 NO_2.

FIGURES 2, 3 and 4 about here

Table 3 presents results of the regression analyses. In all cases the regressions are significant (P<0.0001). The slope coefficients, together with the plots in Figures 2-4, suggest that the positive association between deprivation and air pollution apparent in the 1993 data weakens for all but one of the scenarios investigated. The exception is network development, where the most deprived areas experience the decline in air quality associated with the road building. However, the regression analyses must be treated with caution. Firstly, the normality tests presented in Table 4 indicates that the NO_2 data does not conform strongly to a normal distribution. Several transformations were tested to improve the data but all caused the data to depart further from normality, hence untransformed data were used. Whilst the graphical evidence for inequality appears strong, the moderate departure from normality means that the relationships between deprivation and NO_2 cannot be considered statistically significant on this evidence.

TABLE 3 and 4 about here

To further assess the significance of the apparent inequalities, difference tests were conducted to compare mean NO_2 concentration in the upper and lower quartiles of the deprivation index. The results of these tests (Table 5) show that deprived groups do experience a significantly higher ($P<0.0001$) NO_2 concentration in their residential location than affluent groups. In 1993, the upper, deprived quartile (mean Townsend value of 3.88) experienced a mean annual NO_2 concentration of 33 ug/m^3, compared to a value of 27 ug/m^3 for the more affluent lower quartile (mean Townsend value of -3.05). This pattern occurs for all scenarios, although the extent of the inequality varies. By 2005 the difference in NO_2 concentration between upper and lower Townsend quartiles is predicted to be approximately 2 ug/m^3, a substantial reduction from the 6 ug/m^3 difference observed for 1993. Thus under a do-nothing strategy, environmental inequality reduces significantly.

TABLE 5 about here

Table 6 shows how NO_2 concentration changes in response to modelled transport options. For example, from 1993 to 2005, mean NO_2 concentration is predicted to fall by 30 % in the lower Townsend quartile, but by 36% in the upper, more deprived quartile. In every case, the change in concentration for each transport scenario differs significantly between quartiles. In the case of the do-nothing and road user charge tests, the improvement in air quality is always significantly greater for the deprived quartile than for the affluent quartile. The exception is for network development, where air quality is predicted to fall slightly for everyone (by < 0.1 ug/m^3 NO_2 on average for the city), but the decline will be greatest amongst the most deprived, approximately twice that of the affluent.

TABLE 6 about here

4 Discussion

4.1 Transport strategies and environmental equality in Leeds

The analysis shows that there is social inequity in the distribution of NO_2 in Leeds, with deprived areas experiencing significantly higher atmospheric concentrations than communities of average or above average affluence. The analysis cannot be used to state categorically that deprived communities bear a greater air quality dependent health burden, as other factors determining exposure, discussed above, are ignored. Nevertheless, residential pollutant concentration is routinely used as a proxy for community exposure in air quality health impact assessments, including that used by government (DoH, 1998; WHO, 1995). Thus it is reasonable to conclude that, from a health impact assessment perspective, inequity in respiratory disease burden attributable to ambient air quality also occurs in Leeds.

The analysis also shows that environmental inequity in Leeds is reduced by all but one of the strategic transport options investigated. Under a do nothing strategy, inequity between the most affluent and deprived communities (upper and lower quartiles) declines from 10.6 ug/m^3 in 1993, to 3.7 ug/m^3 in 2005 and just 2.8 ug/m^3 in 2015. These reductions occur as a result of city-wide improvements in air quality, driven by fleet renewal (e.g. more efficient and prevalent emission control technology) that outweighs the effect of forecast growth in total road trips, and act to lower total NO_x emission from the vehicle fleet. Note however, that fine particulate concentrations are forecast to rise in Leeds (and many other UK cities) as emission control technology is insufficient to counteract the expected growth in trips by road transport (Mitchell *et al*., 2003). Thus a do-nothing strategy could not be relied upon to redress inequities in this problematic pollutant, should they occur.

Road user charging also reduces inequity in exposure to NO_2, with the extent of the reduction varying according to the charge option. A Kendall rank test shows that the degree of inequity, measured as the difference between upper and lower quartile NO_2 concentrations (shown in Table 4), is significantly correlated with the change in city-wide annual mean NO_2

concentration ($r_s = 0.87$, $P < 0.01$), and with the change in total PCU-kms ($r_s = 0.87$, $P < 0.01$). In other words, the NO_2 inequity reduces linearly with improvements in city-wide air quality and with reduction in total distance travelled on the road network. The correlation between change in inequity and PCU-kms is clearly not perfect. The double cordon in particular deviates from linearity, not reducing inequity as much as might be expected given the improvement in city-wide air quality it produces. This indicates that, although the reduction in PCU-kms induced by road user charging is the dominant factor reducing inequity, detailed design considerations, such as cordon location, are also significant. From an environmental equity perspective, the effectiveness of road user charging is sensitive to the spatial distribution of socio-economic characteristics; hence the best scheme design may be different for each application city.

Road user charging may be more effective than low emission zones (LEZ's) in addressing environmental inequity. LEZ's are an air quality management tool, currently being considered by UK local government's, in which particular classes of vehicle are barred from an area. A deliberately optimistic analysis has shown that the air quality benefits achieved by LEZ's will anyway occur for the zone in less than five years due to natural fleet renewal (Carslaw and Beevers, 2002). Furthermore, the effect of a LEZ on pollution redistribution outside the zone has not been estimated. Intuitively a LEZ might be expected to lead to an increase in emissions around a zone as the barred vehicles re-route, unless the zone were large enough to suppress trips made by these most polluting vehicles.

The road network developments increased environmental inequity in Leeds, the only transport option investigated to do so. As with the road user charge tests, the change in inequity is a product of changes in PCU-kms travelled and subsequent change in emissions and air quality. The additional road capacity induces an extra 2.4 % trips onto the network, increasing total PCU-kms travelled by 1% (Mitchell *et al.*, 2003). This produces a small (0.3 %) city-wide increase in mean annual NO_2. The effects of this decline in air quality fall disproportionately on the most deprived (Table 6), as the road developments are in the South and East of Leeds, areas which have some of the most deprived communities in the city.

4.2 Methodological Issues in Assessing Environmental Inequity

Through a case study of a medium size English city, a first attempt has been made to assess how an environmental inequity changes over time, and to predict how this inequity varies in response to a series of strategic road transport measures. However, if such analyses are to be useful in achieving a balance between the sustainability meta-goals of economic development, environmental protection and social justice, two key issues must be addressed. These relate to the methods used to assess environmental equity, and how the outcomes of these assessments are interpreted within a justice framework, and incorporated into the planning process.

Bowen (2002) reviewed 42 environmental justice studies conducted sin the USA since the early 1970's, most of which addressed air pollution associated with 'point' sources, such as industrial facilities and landfills. He found that the research body was small and heterogeneous, and the evidence for environmental injustice mixed and inconclusive. The same can be said for the air quality equity studies conducted in the UK (Table 1). Bowen concluded that a principal reason behind the often contradictory conclusions of such research was the lack of consensus on equity assessment methodologies. Several caveats on methods used in the Leeds study are described above, largely addressing the assumptions and difficulties of adequately estimating exposure and health impact arising from transport emissions. However, several other methodological issues are also apparent.

Firstly, deprivation is not automatically the most appropriate demographic measure against which to assess environmental inequity, and other studies stress the importance of different target groups. Stevenson *et al.*, (1999), for example, demonstrated a strong inequity in London air quality, with pollution highest in areas of low car-ownership, a finding which Friends of the Earth used to support their EJ campaign, stating that "traffic pollution is mainly caused by the better off, but the poor feel its effects" (Higman, 1999). Stevenson's finding is also observed for Leeds (Figure 5), perhaps no surprise given that car ownership is a component of the Townsend deprivation index. However, the Friends of the Earth interpretation is suspect, as whilst poorer people do have lower rates of car ownership in general, they do own cars and these are likely to be older and so more polluting. They may also be less well maintained, and represent a higher proportion of the grossly polluting vehicles on the road. Conversely, ownership of older cars is not limited to the poor (e.g. more affluent households may own an older second car), and affluent people travel further.

FIGURE 5 about here

Such factors were considered by Mitchell and Dorling (2003) in an analysis of all census wards in Britain. The results showed that affluent British households do not emit significantly more NO_X from private cars than poor households. However, a very significant inequity was observed, in that a small proportion of census wards were characterised as emitting the least NO_X, but were wards with the highest NO_2 concentrations. Furthermore, the population within these wards were amongst the poorest in Britain, and so are the least able to address the impacts of the pollution that they do not contribute to (e.g. relocate). This appears to be a gross injustice, and suggests that EJ analyses that solely addresses deprived or minority groups, as recommended to Federal agencies in the USA for example (US EPA, 1998; US EPA 1999), may be flawed. If justice, rather than simply equity concerns are to be addressed, then Feitelson's (2002) recommendation to focus analysis on comparisons of the attributes of users of transport systems to those affected by such systems, appears sound.

A second methodological issue is the appropriate treatment of scale. Demographic data in this study related to SURPOP grid cells, each 4 Ha in area. However, larger units, postcodes or census areas for example, could be expected to generate lower inequalities as the extremes in the data are lost. Such units may introduce problems in transport equity studies, as their boundaries are often delineated by transport routes. The geographic area of analysis is also important, as its selection may act to artificially inflate or dilute the representation of the target demographic group (US EPA, 1999), and is critical to the identification of an appropriate control group. In general, scale issues in transport environmental equity analysis present greater challenges than for single site analysis, as whole transport systems may need to be considered.

A third methodological issue relates to the criteria used to define injustice. Cutter's (1995) definition of environmental justice is one where target (e.g. minority) groups share the same burden of environmental harm as the rest of the population. In contrast, the criterion adopted by US Federal agencies is one where target groups do not bear a disproportionately higher adverse human health or environmental impact of a policy, programme or activity. Impacts are considered unjust if they 'appreciably exceed' impacts to the general population, or are above 'generally accepted norms', but this interpretation is left to the judgement of the analyst (Wilkinson, 1998; US EPA, 1998).

What constitutes an adverse impact is also relevant, and compliance with environmental standards may not always be appropriate. For example, in the case of the Leeds study, an inequity in NO_2 is apparent, with deprived groups exposed to NO_2 concentrations that are higher than in the general population: by 23% in 1993, and 12% in 2005. The 1993 inequity may be considered an injustice, as then there were many failures of the NO_2 air quality standard. For 2005, full compliance is forecast, thus the predicted inequity may be considered acceptable. However, recommended air quality dose-respiratory disease response relationships are linear, with no lower effect thresholds for long term exposure (DoH, 1998; WHO, 1995), hence a health impact is feasible at levels below the standard. Also, air quality standards are set with reference to health impacts that are considered economically acceptable. Thus the use of environmental standards in defining when an inequity is unjust should not be accepted uncritically. Note that target groups (e.g. children) may also be more sensitive to pollutants than the general population, but that standards are set with reference to the general population.

Finally, ensuring that equity analyses are adequate in scope presents substantive practical challenges. This is particularly evident with respect to transport systems, where multiple and cumulative environmental impacts occur, and where indirect effects can be significant. The Leeds analysis, for example, ignores emissions of other pollutants that affect health, as well as noise and severance. Indirect effects, such as the impact of strategic transport developments on future land use and urban structure are also neglected. The SPARTACUS system (LT *et al.*, 1998) is the most advanced in this respect, addressing several environmental impact criteria, and second order impacts via its land use transport interaction model.

Methods of environmental equity and justice analysis are generally poorly developed, but are evolving. In the US, general advice on the broad approach to EJ analysis is available (EPA 1998; EPA 1999), and is being refined through EPA enforcement actions and civil lawsuits (Wilkinson, 1998). However, whilst the Presidential memorandum accompanying Executive order 12898 requires EJ reviews of proposed federal legislation and regulation affecting air quality, the focus of EJ analyses remains individual facilities and projects rather than plans and policies (Warner, 2002), although there are some examples of the latter in the transportation field (FHA, 2000). Feitelson (2002) comments that where transport infrastructure is concerned, a public transport terminal for example, equity analysis is more complex than with those hazardous facilities (factories, landfills etc.) more usually subject to equity analysis, as those most impacted upon may benefit preferentially from their proximity to the service. There remains then, much scope for developing methods that are capable of assessing policy impacts on environmental inequity, which are robust when applied to transport systems, and which have relevance to communities outside the USA.

4.3 Addressing Environmental Equity in the Planning Process

The inclusion of environmental equity assessment into the planning process should lead to the promotion of social justice and a greater balance between the three meta-goals of sustainable development. However, as noted in the introduction, environmental equity is not equivalent to environment justice, and issues of causality and interpretation of 'what is fair' also need to be addressed. Issues of causality can engender different assessments of justice, as environmental inequities can arise in different ways. Concerns over discrimination within the planning process was an original driving force of the US EJ movement, and if demonstrated, would likely lead to the conclusion that observed inequities are unjust. In contrast, concluding that the cause of an environmental inequity is a neighbourhood transition process, where inequities are the product of individual choice and the distribution of wealth, income and market forces (Freeman, 1972), may well lead to the conclusion that there no injustice exists. Understanding how environmental inequities arise is therefore relevant to their interpretation within a justice framework, and also to understanding how policies and plans modify environmental equity patterns. Feitelson (2002) warns that identifying whether past transport policies have been systematically biased will be difficult, due to a long series of assumptions and interpretations when assessing equity, and hence that research in this area should adopt a case study approach, focusing on the decision making process.

Assessment of environmental equity patterns under alternative concepts of justice is rare, but not without precedent. The SPARTACUS project (LT *et al.*, 1998), modelled the environmental impacts associated with a series of urban transport planning options, and assessed the outputs against alternative justice theories. The theories included a utilitarian approach where average benefits are maximised, an equal sharing of impacts approach, an egalitarian approach in which inequalities are reduced, and a Rawlsian approach where the absolute welfare of the least well off is emphasised. Application of the different justice theories gave rise to different preferred transport options.

There are clearly several key issues to be addressed if EJ is to be effectively addressed in the planning process. These include: defining appropriate procedures and analytical methods of environmental equity assessment; predicting how equity changes over time, and in response to policies and plans; and how to interpret inequities within a justice framework (including causality and 'what is fair' issues). There is also the wider challenge of balancing justice concerns against economic development and environmental protection goals.

5 Conclusions

This study aimed to determine if environmental inequity occurs with respect to air quality in Leeds, and if so, to what extent strategic transport measures of current interest to local government's might alter this pattern. Inequity in residential NO_2 concentration in Leeds does occur, and for the 1993 base year was substantial, and likely to contribute to above average respiratory disease burden in deprived communities. However, the analysis shows that even with no intervention, the observed inequity is in decline, as fleet renewal results in lower total emissions from road transport, and an improvement in air quality which all benefit from, but which deprived communities benefit most from.

Perhaps contrary to expectations, road user charging is capable of promoting environmental equity. This should be welcome news to local governments in the UK, who now have legal powers to implement road user charging to control congestion and pollution, but who have expressed concerns about the impact that schemes may have on the redistribution of traffic and pollution (DETR, 1998b). Nevertheless, detailed design issues still require careful assessment for each city where a charge is considered.

Overall, change in environmental equity was strongly related to change in total PCU-km travelled on the network, total emissions and city-wide air quality, and any systemic measure that improves air quality is likely to be beneficial in equity terms. Thus promoting emission control technology and clean fuels, encouraging a modal shift from private to public transport, and limiting travel demand are all likely to be beneficial in equity terms. Perhaps more importantly, the analysis indicates that goals of environmental protection and social justice are complementary, at least with respect to air quality, and hence addressing these goals simultaneously does not restrict the range of options available to address these objectives. This does, however, remain to be tested more widely.

From a sustainability perspective, justice issues have yet to receive the attention they deserve, and demands to address environmental equity issues in the planning process are growing. The pressures for EJ in Europe are driven by the same forces as those that occurred in the USA: grass roots activism (e.g. Pennycook *et al.*, 2001; FoE, 2001; Dunion, 2003); evidence of inequity (Dolk *et al.*, 1998; Elliot *et al.*, 2001, see also Table 1), and high level policy initiatives (e.g. UN ECE, 1999) although litigation has not been a significant factor. Whilst environmental equity assessment can support the promotion of social justice in transport planning, considerable further research is required as there are few, if any, generic conclusions to guide policy makers.

In the case of the Leeds study the analysis could usefully be extended to address: more pollutant variables (especially PM_{10}, a significant current health concern); alternative outcome measures (e.g. health impact); and other target groups (e.g. ethnic minorities, transport system users and non-users). There is also scope for improved assessment of exposure, for addressing a wider range of transport and non-transport measures affecting urban air quality, and for assessing the generality of the findings reported here, through extension to other cities. There are however, a series of more fundamental questions to be addressed before environmental equity analyses becomes a more widely accepted tool to guide sustainable transport planning. These relate to: the development of appropriate procedures and methods for environmental equity assessment, including equity responses to transport policies and plans; appropriate mechanisms for interpreting inequities within a social justice framework; and procedures for balancing these results with economic and environmental goals. These issues cannot be addressed fully without first adequately involving the public, the ultimate arbiters of environmental justice.

The importance of public participation is recognised in EPA guidance on EJ assessment, including measures that require assessment under the Clean Air Act (EPA, 1998; EPA, 1999). The guidance addresses issues of community involvement in scoping (e.g. identifying target groups and preferred mitigation measures), and in reviewing the EJ assessment. By extending public involvement to the other key issues described above, more robust assessments would result. However, from their survey of environmental concerns in disadvantaged communities, Burningham and Thrush (2001) found that air quality was not a major concern despite the fact that asthma suffers who were not car owners recognised a connection between traffic, air pollution and their ill health. This lack of concern was attributed to respondents' perception that air quality was inextricably linked to continual traffic growth, which they perceived as an intractable problem. This study therefore suggests that encouraging key groups to participate in environmental equity and justice evaluations may be difficult.

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Figure 1. Annual mean NO_2 in Leeds for 2005 under the do nothing option

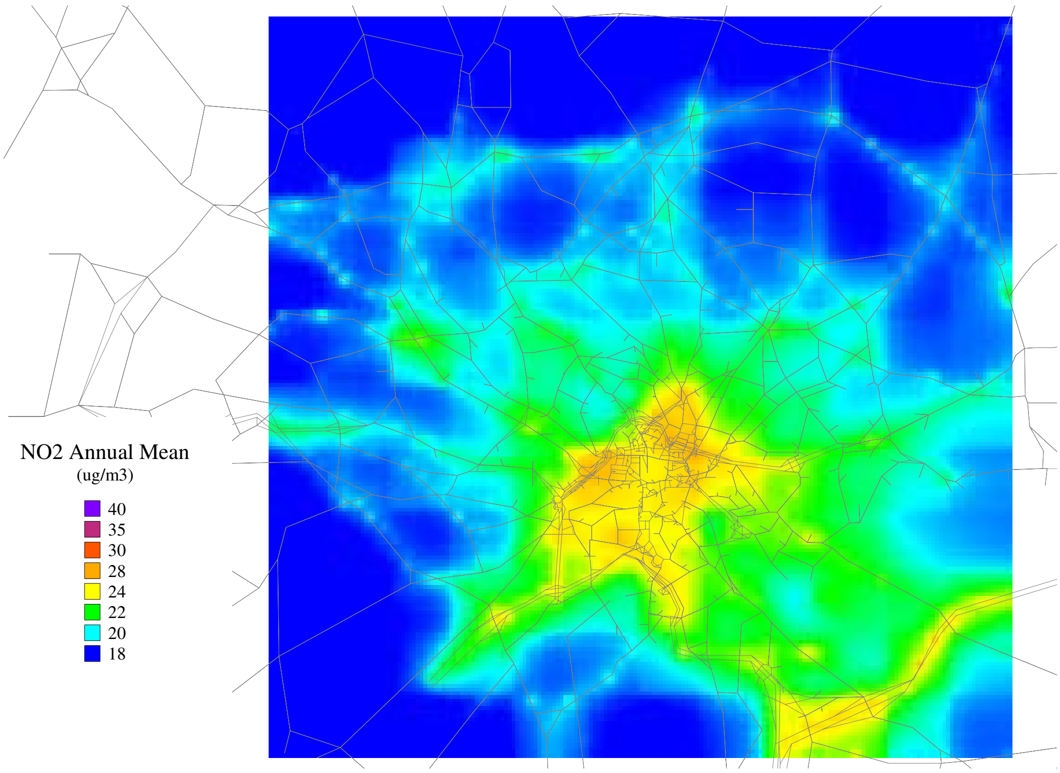


Figure 2. Environmental equity under the do nothing scenario

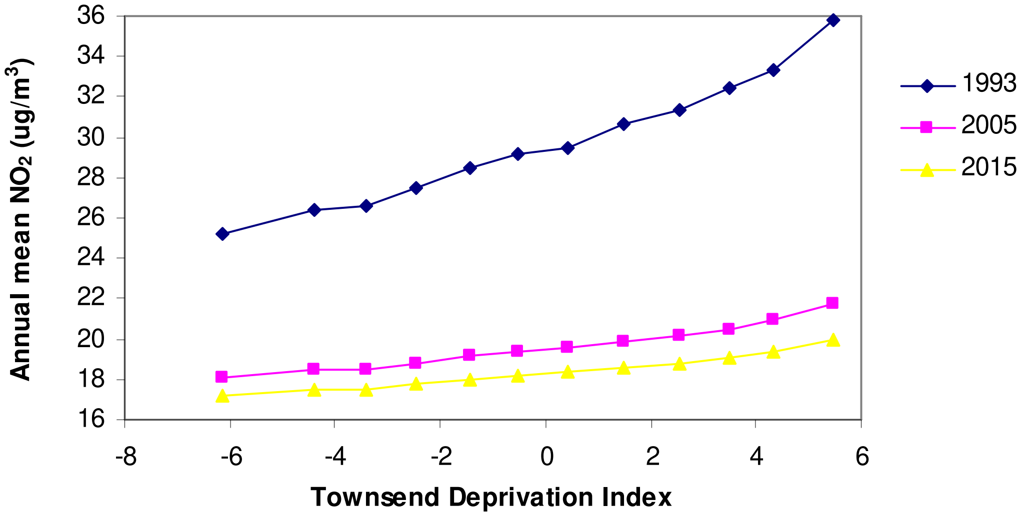


Figure 3. Environmental equity under the road network development scenario

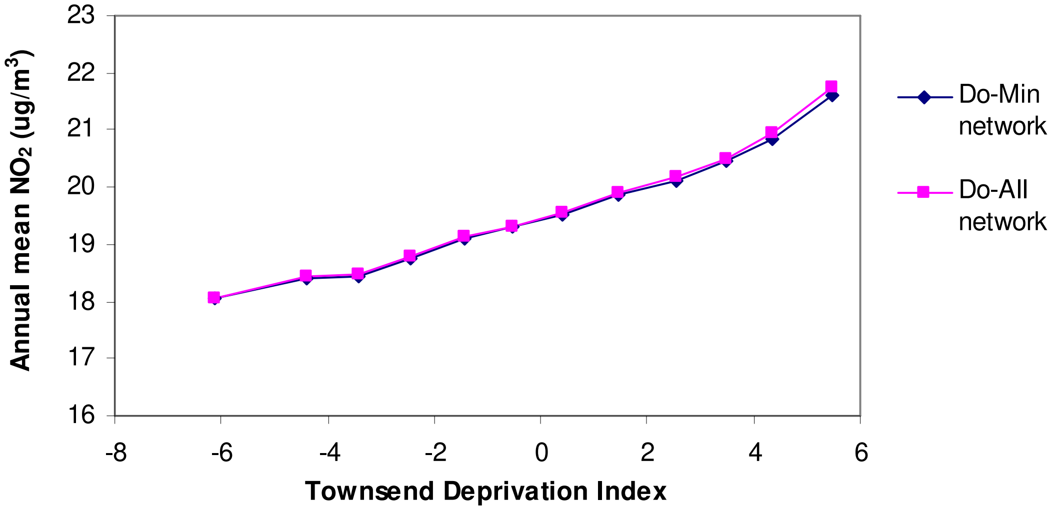
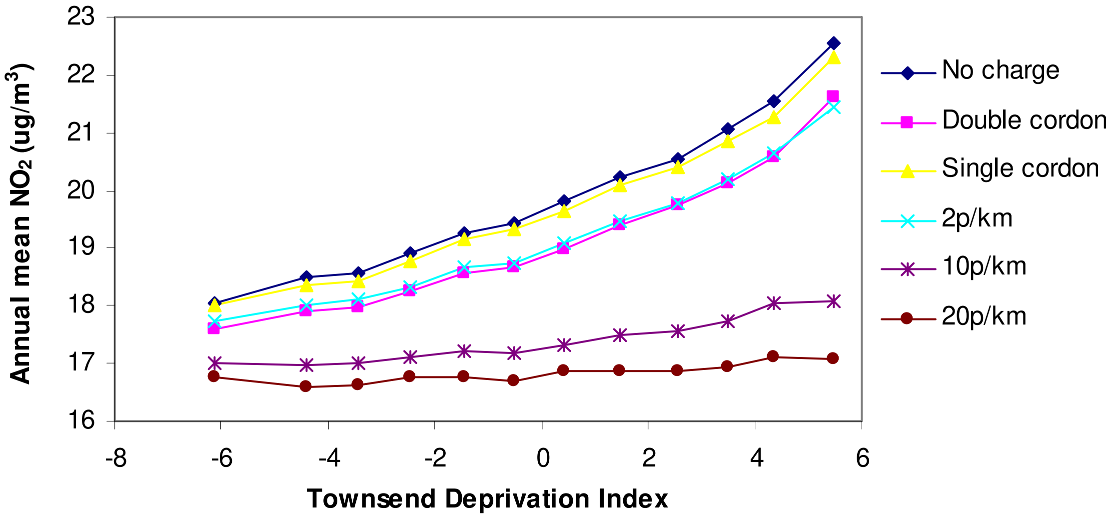
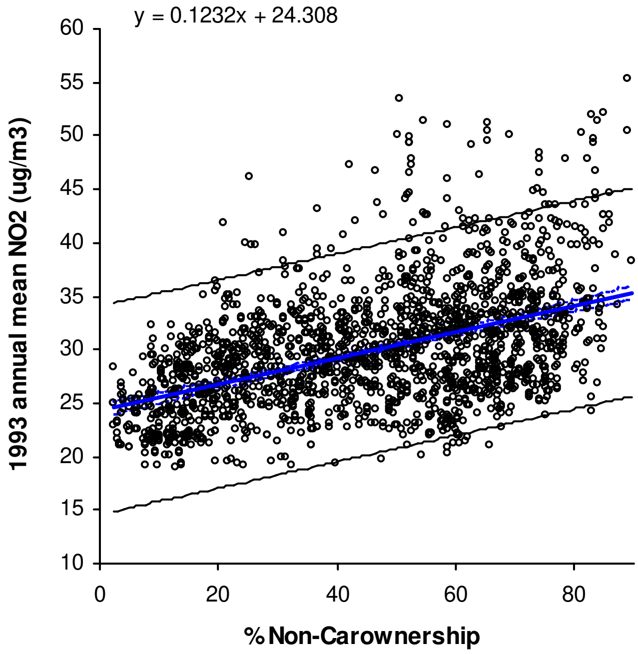


Figure 4. Environmental equity under road user charge scenarios



Notes: 1.Higher Townsend value indicates greater relative poverty; 2.Total N=1851.

Figure 5. Modelled NO_2 concentration and observed car-ownership in Leeds.



Notes:

1. Car ownership data for 1991, NO_2 data for 1993;
2. Bars denote 95% confidence interval;
3. Residuals in non-car ownership are normally distributed;
4. N=1851, P<0.0001.

Table 1. UK air quality social-equity studies

Reference	Study location	Socio-economic indicator	Observed association with socio-economic indicator
Stevenson *et al.*, 1998; 1999	Wards in Greater London	Poverty: Income; Car ownership.	A positive association between deprivation and NO_2 and respiratory diseases.
King and Stedman, 2000	Wards in 5 UK cities	Poverty: various deprivation indices	Weak positive correlation of NO_2 and PM_{10} with deprivation for London, Belfast and Birmingham, but the inverse for Glasgow and Port Talbot (PM_{10}).
McLeod *et al.*, 2000	Local authority districts	Ethnicity: % of household heads from India & New Commonwealth)	A positive association with NO_2, SO_2 and PM_{10}, not attributed to multi-collinearity with deprivation measure.
		Poverty: Social class index	A weak positive association with PM_{10} and SO_2; Very weak positive association with NO_2. A negative association with NO_2 and SO_2 when population density accounted for.
Pennycook *et al.*,. 2001	Wards in Bradford	Poverty: Index of multiple deprivation	Mapped data suggests that NO_2 and PM_{10} "tends to be highest in the most deprived areas".
Pye *et al.*, 2002	Wards in four large UK cities.	Poverty: Index of multiple deprivation	A weak positive association with NO_2 and PM_{10} in London, Birmingham and Belfast but no association found for Cardiff.
Lyons *et al.*, 2002	Glamorgan, Wales	Poverty: Social class	No association with NO_2, but a very small sample (171 adults)
Brainard *et al.*, 2002	Enumeration districts in Birmingham	Ethnicity: % self reporting as white, Asian or black	A strong positive relationship with ethnicity but difficult to separate effect from poverty.
		Poverty: Various indexes	A strong positive relationship with poverty, but difficult to separate effect from ethnicity.
		Age: E>60, Γ >65 years; all <15 years	No association with NO_2 or CO emission for any age group.
Mitchell and Dorling 2003	All census wards in Britain	Poverty: Breadline Britain index	Poorest wards emit least NO_X from resident vehicles but have highest NO_2 exposure.
		Age: <1 year, then all ages in 5 year age bands	NO_2 40-80% above mean for young children and 18-40 yr olds, reflecting urban to rural life stage migration.

Table 2. Spatial redistribution of modelled annual mean NO_2 concentrations under alternative transport scenarios in Leeds.

Scenario [1]	**NO_2 annual mean**		**NO_2 99.8 percentile**	
	No. sites degraded [2]	No. sites improved [2]	No. sites degraded [2]	No. sites improved [2]
Do Nothing				
1993-05	10	3584	2	3597
2005-15	0	3600	0	1994
1993-2015	0	3600	1	3598
Road network development				
Do-All	399	104	44	14
Road User Charging				
Single cordon (£3)	231	824	47	131
Double cordon (£1 + £2)	7	3535	10	452
2p/km distance charge	10	3222	8	348
10p/km distance charge	0	3599	0	987
20p/km distance charge	0	3600	0	1275

1. All scenarios are for 2005 unless otherwise stated.

2. Number of sites where NO_2 has changed by > 1% from reference scenario. 3600 sites were modelled, each representing air quality for a 4 ha grid cell.

Table 3. Regression of modelled NO_2 concentration and observed deprivation in Leeds.

Transport Scenario	R^2	F stat.	Intercept	Slope	Slope 95 % confidence interval
Do Nothing					
1993	0.16	341.9*	29.57	0.8205	0.734-0.908
2005	0.17	387.4*	19.48	0.2846	0.256-0.313
2015	0.14	293.3*	16.89	0.1224	0.108-0.136
Network development.					
Do All (2005)	0.17	380.2*	19.53	0.2903	0.261-0.319
Do All (2015)	0.18	409.5*	18.32	0.2234	0.202-0.245
Road User Charging					
No charge	0.19	420.3*	19.79	0.3599	0.352-0.394
Single cordon (£3)	0.18	412.3*	19.65	0.3453	0.312-0.379
Double cordon (£1+£2)	0.18	408.7*	19.02	0.3162	0.286-0.348
2p/km distance charge	0.18	396.3*	19.11	0.3067	0.276-0.337
10p/km distance charge	0.11	218.0*	17.36	0.1065	0.092-0.128
20p/km distance charge	0.03	60.7*	16.81	0.0415	0.031-0.052

All scenarios are for 2005 unless otherwise stated. * P< 0.0001.

Table 4. Normality tests of modelled NO_2 concentration in Leeds.

Scenario	Skewness	Kurtosis	Shapiro-Wilk
Do Nothing			
1993	0.95	11.17	0.952*
2005	1.15	12.80	0.933*
2015	1.43	14.14	0.892*
Network development.			
Do All (2005)	1.20	13.12	0.928*
Do All (2015)	1.21	13.07	0.926*
Road User Charging			
No charge	1.43	15.57	0.903*
Single cordon (£3)	1.37	14.58	0.907*
Double cordon (£1+£2)	1.57	17.18	0.884*
2p/km distance charge	1.54	16.65	0.888*
10p/km distance charge	1.72	18.94	0.862*
20p/km distance charge	1.89	23.93	0.838*

* P< 0.0001

Table 5. Tests of difference in Leeds NO_2 (ug/m^3) in the upper and lower quartiles of the Townsend index.

Scenario	Mean annual NO_2 (ug/m^3)		2 tailed T-Statistic [2]
	Lower Townsend Quartile [1]	Upper Townsend Quartile [1]	
Do Nothing			
1993	26.99	32.95	16.48*
2005	18.60	20.65	17.70*
2015	16.50	17.38	15.79*
Network development.			
Do All (2005)	18.63	20.73	17.53*
Do All (2015)	17.63	19.24	18.22*
Road User Charging			
No charge	18.72	21.29	18.11*
Single cordon (£3)	18.61	21.09	18.18*
Double cordon (£1+£2)	18.11	20.37	17.84*
2p/km distance charge	18.20	20.40	19.92*
10p/km distance charge	17.06	17.84	13.89*
20p/km distance charge	16.68	16.99	8.28*

1. Mean Townsend deprivation score in lower quartile is -3.05, and 3.88 in the upper quartile.

2. There are 463 degrees of freedom, and * $P < 0.0001$.

Table 6. Tests of difference in Leeds NO_2 (% change by scenario) in the upper and lower quartiles of the Townsend index.

Scenarios compared	Change in annual mean NO_2 (%)		2 tailed T-Statistic [2]
	Lower Townsend Quartile [1]	Upper Townsend Quartile [1]	
Do Nothing			
1993-05	-30.01	-36.37	15.40*
2005-15	-11.05	-15.49	17.08*
1993-2015	-37.51	-46.02	16.26*
Road network development			
Do-All	+0.18	+0.34	3.33^{+}
Road User Charging			
Single cordon (£3)	-0.57	-0.89	3.61^{+}
Double cordon (£1 + £2)	-3.18	-4.26	8.67*
2p/km distance charge	-2.67	-4.10	13.22*
10p/km distance charge	-8.50	-15.54	18.38*
20p/km distance charge	-10.46	-19.26	18.60*
Single - double cordon	-2.62	-3.39	6.18*
Double cordon - 2p/km	-0.51	-0.15	-2.66^{++}

1. Quartile mean Townsend scores as Table 5.
2. There are 463 degrees of freedom, and * $P < 0.0001$; $^{+}$ $P<0.001$; $^{++}$ $P<0.01$.