CONCEPTUAL FRAMEWORK FOR ANALYSING THE DISTRIBUTIVE IMPACTS OF ENVIRONMENTAL POLICIES

David PEARCE

Prepared for the OECD Environment Directorate Workshop
On The Distribution of Benefits and Costs of Environmental Policies
PARIS March 2003

Revised and final copy

April 2003

Contact: d.pearce@ucl.ac.uk

CONTENTS

GLOSSARY of concepts

As several technical concepts are used in this report, and after being introduced are summarised by a single statistic, this glossary summarises them.

Income elasticity of demand

This is the change in the quantity demanded of some environmental asset in response to a small change in income. It is given by the formula:

$\eta = \partial E.Y/\partial Y.E$

where E is the quantity demanded, Y is income.

Income elasticity of willingness to pay

This is the change in the willingness to pay for some environmental asset in response to a change in income. It is given by the formula:

$\omega = \partial WTP.Y/\partial Y.WTP$

where WTP is willingness to pay.

Price elasticity of demand

The price elasticity of demand is the change in the quantity demanded with respect to a change in the price of the environmental asset. It is given by the formula:

$p = \partial E.P/\partial P.E$

where P is price and p is the price elasticity.

Under certain circumstances:

$\omega = \eta/p$

Income elasticity of pollution

Some authors use the notion of an 'income elasticity of pollution' which is defined as the percentage change in pollution for a given percentage change in income. Note that, despite the similarity in appearance, this is not the same as the income elasticity of demand, but refers more to the supply of pollution at different income levels. Clearly, however, the notion could reflect demand forces.

$d = \partial E_S.Y/\partial Y.E_S$

where E_S is the ambient concentration of pollution.

1 THE ISSUE

A sizeable empirical literature exists on the relationship between environmental quality and socio-economic groups within a nation's borders[1]. The hypothesis tested by this literature is that environmental quality is regressively distributed across socio-economic groups, i.e. low income groups are exposed to higher environmental risks than high income groups. If this is true, and if the distribution across income groups is not freely chosen by those groups, then an issue of distributive equity arises. Regressive distributions could be deliberately chosen: it may be that low income groups have a lower demand for environmental quality than high income groups. Alternatively, higher levels of pollution may be associated with associated benefits - e.g. lower property prices - that compensate those groups for higher environmental risk. But regressive distributions may also be the result of an unequal endowment of political power and limited ability to adjust to environmental risks. In so far as unequal political power explains the regressivity, an equity issue still arises. Even when power is fairly equally distributed, the public good nature of many environmental goods, and hence the public bad nature of the risks, may produced compromise allocations of the good that under-supply the good to higher income groups and over-supply it to low income groups, still producing an equity problem. These alternative explanations for regressivity of risks are explored in detail in Section 3.

From a policy standpoint, equity is a goal of social and economic policy in OECD countries. What constitutes 'equity' is not straightforward and the issue is not investigated in any detail further here (for excellent treatments see Young (1994) and Zajac (1995)[2]. In the current context, an equitable outcome is taken to be one that produces an equal exposure to environmental harm, or the equal per capita endowment of environmental benefits, exposure and endowments being measured across income groups. Policy goals may then be formulated in terms of reducing inequality in exposure to harm, or increasing equality in the endowment of benefits. Both goals characterise the movement for environmental equity, and more popularly known as environmental justice that has assumed some importance in policy discussions in the USA and is now being discussed in European countries[3]. Unfortunately, what is meant by 'harm' and 'benefit' is itself not straightforward, an issue explored further in Section 3.

---

[1] The cross-national literature is ignored here. The relevant empirical literature is encapsulated in the notion of an 'environmental Kuznets curve' (EKC) which traces out relationships between environmental quality (or resource use) and real income per capita. While it is usually characterised as taking on the shape of an 'inverted U' curve - with environmental degradation or resource use at first rising with income growth and then falling - the empirical evidence is in fact more ambiguous than is generally supposed. See Harbaugh et al. (2000). The EKC function usually takes the form $E_{it} = a + b.Y_{it}/POP_{it} + c.(Y_{it}/POP_{it})^2 + \varepsilon$, where E is environmental degradation, Y is real income, POP is population, ε is an error term, i is location, t is time, and a,b,c are parameters to be estimated. Note that we also ignore the literature that deals with distribution of environmental goods across 'economic' groups - consumers and producers for example.

[2] An excellent classification of equity concepts can also be found in Carraro (2002).

[3] There are many strands to the EJ literature and they extend way beyond the empirical investigation of the social incidence of environmental costs and benefits. One relates to the ethical underpinnings of the techniques used to assess environmental justice. A significant issue, for example, is whether individuals have some natural right to a 'clean' or 'zero risk' environment and, if so, what status is then appropriate for procedures such as risk assessment in which risks are explicitly traded against costs of risk reduction and against non-environmental benefits. This literature is not explored in this report other than tangentially. For extensive discussion see the special issue of *Human and Ecological Risk Assessment*, Volume 6, No.4, 2000, especially the papers by Goldman (2000), Sexton

A second reason for being concerned with unequal distributions of harm is that they may be inefficient. This point is less obvious that the equity issue. Consider Figure 1 which shows a stylised linear dose-response function linking an environmental indicator, say ambient pollution concentration, C, to a 'damage' indicator, D (health damage, ecosystem damage etc.). Assume that the rich occupy land where C = 10 and the poor occupy land where C = 40. The average C is then C = 25. Total damage is then given by D = 1 for the rich and D = 4 for the poor, i.e. total D = 5. Equalising risks involves each group moving to locations where C = 25. Each then faces damage of D = 2.5 so that total damage is unchanged at D = 5. Now assume the dose-response function is non-linear and convex. Figure 1 shows that D for the rich is 1.2 and D for the poor is 8.0, making total damage 9.2. If risks are now equalised at C = 25, each group faces damage of 3.5, giving a total damage of 7. Aggregate damage is therefore reduced by equalising risks if dose response functions are non-linear. This is the efficiency argument for equalising the distributive incidence of environmental hazards.

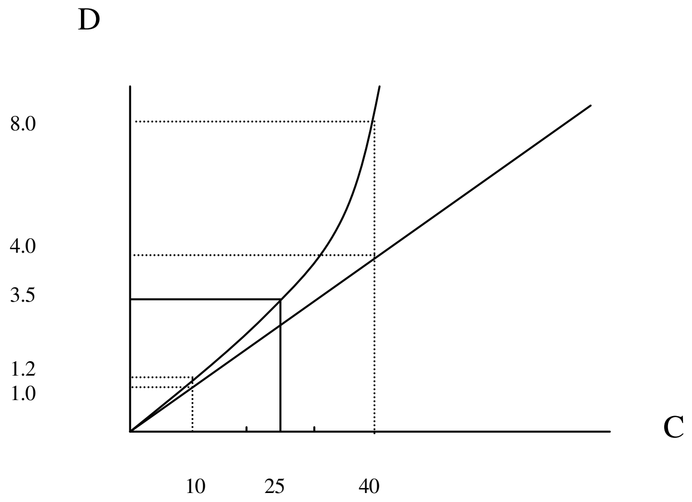

Figure 1: Efficiency and dose-response functions

The purpose of this report is to provide a general conceptual framework for the analysis of the socio-economic distribution of environmental bads and goods[4]. This framework requires not only the setting up of hypotheses about the existing distribution of environmental quality, but also a model to explain why that distribution comes about. The explanatory model is essential since the focus on regressivity (or otherwise) alone can be misleading if there are offsetting factors that compensate for the inequality of risks, or if inequality is freely chosen. Using this framework we then analyse the available empirical literature to see what evidence can be adduced for the hypotheses that the existing distribution of environmental bads is regressive - the 'environmental injustice hypothesis'. Second, we look at the evidence on the distribution of benefits from environmental policies. In this case there are no *a priori* expectations about the distribution of benefits: they may be progressive or regressive. Third, we investigate the

(2000), Simon (2000) and Foreman (2000). It should also be borne in mind that the EJ literature is itself a sub-set of a larger literature that analyses inequalities in health and access to public goods generally, the argument being that poorer people tend to have lower health status, lower life expectancy etc. This substantial literature is not considered here. For a review see Wagstaff and van Doorslaer (2000). The *British Medical Journal* (http://bmj.com) has carried a large number of articles on health and inequality, as has the *Journal of Policy Analysis and Management*.

[4] A 'bad' is the opposite of a good, so that pollution would be regarded as a bad, and pollution reduction as a good.

relevance of accounting for the distribution of cost burdens along with the distribution of benefits of policies - the issue is whether it is the gross or net benefit (benefit minus cost) incidence that matters for policy. Fourth, we look at the issue of distribution from another perspective, namely whether the demand for environmental quality is 'income elastic' or not. If it is income elastic, the suggestion would be that improvements in environmental quality will tend to be biased towards higher income groups since they demand more of it. Meeting varying demands for environmental quality is, of course, a sign of an efficient policy. But an equity issue may still arise of those demands are affected by the prevailing distribution of income. The argument may then be about whether the prevailing distribution of income is itself fair, or, if judged fair, whether there are some overriding moral principles that require an equal distribution of environmental quality: i.e. income distribution could be fair but environmental quality distribution unfair. Finally, policy issues are addressed with the aim of seeing how far pursuit of equality of environmental endowments is consistent with efficient environmental and economic goals.

2 THE OECD PROGRAMME

The current report is part of OECD's wider programme of work on *The Social and Environmental Interface: Enhancing the Quality of Life*, a priority action under OECD's Environment Strategy for the First Decade of the 21st Century. Work components under the activity Social and Environmental Policy Integration include:

Environment and employment
Distribution of environmental quality
Distributive effects of selected environmental policies
Environmental awareness and communication strategies

The current report addresses the second and third issues.

3 ENVIRONMENTAL JUSTICE AND ENVIRONMENTAL POLICY

The notion of environmental justice (EJ) is not straightforward. Roberts (2000) follows US EPA (1998) in defining EJ as:

> ' [the] fair and equitable treatment of all people, regardless of race, ethnicity, income, national origin, or educational level in the development and implementation of environmental laws, regulations and policies. (Roberts, 2000, p.537).

This definition tends to focus on the 'equal' impact of policy, whereas much of the EJ movement is concerned with not just this aspect of policy, but with the correction of any existing inequality of risks between income and racial groups. The only OECD country in which concern for environmental justice is mandatory is the USA. Other countries, however, provide guidelines on assessing the distributional impacts of policy measures and some of these guidelines have strong official backing. OECD (2002) summarises the policy initiatives.

<u>The USA</u>

Bowen (2002) traces the political history of the EJ movement in the USA, and notes that several empirical studies had a major influence on succeeding legislation - notably those by Bullard

(1983), US GAO (1983) and the United Church of Christ (1987)[5]. Race as well as income was a major concern in these studies, the argument being that ethnic minorities are also unfairly exposed to environmental risks, in each case landfill sites with hazardous wastes. A further significant impetus to legislation were the 'NLJ articles' (Lavelle and Coyle, 1992), a set of articles in the *National Law Journal* that charged US EPA with discrimination against minorities and low income areas when prosecuting violations of environmental law. It was alleged that lower penalties were applied to these areas, suggesting that EPA provided less environmental protection to them than to richer, white areas[6].

In 1990, the US EPA established an internal working group to study the links between minority and low income populations and environmental hazards. In 1992 the EPA established an Office of Environmental Equity to investigate EJ concerns and in the same year published a major report on the issue (US EPA, 1992). Formal EPA guidance was issued in 1998 (US EPA, 1998). In 1994 President Clinton enacted *Executive Order 12898* which requires federal agencies formally to address issues of environmental hazards in low income and minority communities. EO 12898 requires federal agencies to develop environmental justice strategies. Programmes, policies, planning and public participation procedures should be revised to:

> '…promote enforcement of all health and environmental statutes in areas with minority populations and low income populations; ensure greater public participation; improve research and data collection relating to the health and environment of minority populations and low income populations; and identify differential patterns of consumption of natural resources among minority populations and low income populations.' (EO 12898).

<u>Other OECD</u>

The United Kingdom has formalised central government advice on the treatment of different income groups in policy appraisal (HM Treasury, 2002). Any policy option must be analysed to determine whether impacts differ by socio-economic groupings. Depending on the judged significance of the distributional incidence, action may be required to modify the policy in question. Where formal analysis is called for, 'distributional weights' may be employed (for the theory and empirical illustrations see Sections 9.1 and 9.2 of this report). Such weights raise the social value of any unit monetary gain to low income groups relative to other, richer groups. Such distributional weights can then be incorporated into a cost-benefit appraisal of the policy, this form of appraisal being that generally recommended for government policy. Annex 6 of the Treasury Guidance provides more detail on the size of the weights to be adopted. Thus, those in the lowest quintile of relevant income would have their gains (or losses) multiplied by a factor of 1.8 relative to those on average income (these multiples hold for the value of the 'elasticity of marginal utility of income' of unity - see Sections 9.1 and 9.2 below). Those in the highest quintile would have their gains/losses multiplied by a factor of 0.4. The Guidance further notes that, where the correction of social inequality is an explicit aim of policy, then the resulting

---

[5] Bowen (2002) regards all these studies as being of comparatively low scientific quality.

[6] The 'NLJ articles' are themselves the subject of a detailed debate as to the validity of the allegations - see Ringquist (1998) and Atlas (2002). Much of the debate relates to the source of information - EPA's Civil Enforcement DOCKET database - and the ways in which researchers have interpreted it. Ringquist suggests the NLJ allegations are false. Atlas also finds serious fault with the NLJ analysis but additionally suggests Ringquist's results are invalid due to misuse of the DOCKET database.

weighted impacts can be further weighted to reflect the judgement that an extra unit of wellbeing to a low income group is more valuable than that of a higher income group.

In terms of policy, a number of OECD countries seek to make allowance for the incidence of environmental policy costs on low income households. These allowances tend to reflect the judgement, sometimes back by statistical evidence, that lower income households have higher expenditures, proportionate to their incomes, for environmental services. Put another way round, some environmental expenditures constitute a higher proportion of low income households' income compared to high income households. Procedures used include establishing a 'consumption floor' below which no tax is levied, and having rising tariffs for consumption of the goods in question, effectively producing a cross-subsidy from richer to poorer groups. Examples of such measures are: the Climate Change Levy in the UK which is not applied to households at all, and a lower rate of VAT on household energy bills than on other VATable items; Germany where a 50% rebate is provided on electricity taxes for storage heaters which tend to be used by low income households; the Netherlands where there are exemptions for some low income households from waste collection and sewerage charges (de Kam, 2002). Other examples are given in OECD (2002) and in de Kam (2002).

4 CONCEPTUAL FRAMEWORK

4.1 The EJ hypotheses

The hypotheses to be tested are (a) that the existing distribution of environmental 'bads' is regressive across income groups[7], and (b) environmental policy is distributionally biased against low income groups. Hypothesis (a) probably more fairly describes the concerns of the EJ movement, but some of the literature is also concerned with hypothesis (b). As far as the first hypothesis is concerned, the expectation is that environmental risk, ER, (which is to be defined shortly) declines with real per capita income (Y/N), as shown in the stylised function in Figure 2. Figure 2 is consistent with the 'environmental Kuznets curve' (EKC) referred to earlier (see footnote 1) and which is typically estimated across different nations. Figure 3 shows the full EKC that is usually postulated in the cross-country literature, and serves to highlight an immediate methodological issue. In Figure 3, the level of environmental risk ER* is seen to be consistent with two different income levels Y1/N and Y2/N. Thus. 'rich' and 'poor' could be exposed to the same level of risk, implying that there is no problem of environmental equity. However, as the Y1 group improve their incomes so ER for them rises, while for the Y2 group it declines. Observation of equal risk exposure is not therefore sufficient to establish that there is no environmental inequity. The rate at which risk changes with respect to income change also matters. Very often, this two-part test is not carried out in the empirical literature.

[7] For the rest of the report we confine attention mainly to income groups, but a significant part of the literature is concerned with racial groups as well. In so far as race and income tend to be correlated, it can usually be taken that findings related to income also hold for racial groupings, but not always.

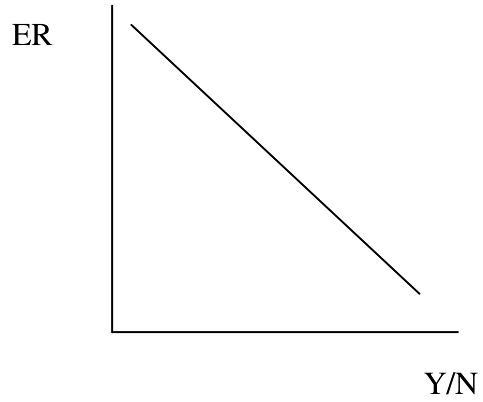

Figure 2: Income and Environmental Quality

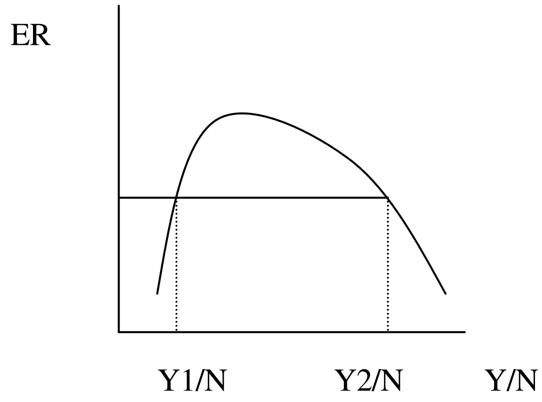

Figure 3: Income and Environmental Quality

4.2 The risk unit of account

The second methodological problem relates to the measure of environmental risk, ER. A distinction can be made between 'physical' risk measures and 'perceived' or 'preference-based' measures of risk. Physical risk measures tend to relate to some indicator of pollution or hazard such as ambient pollution concentrations or proximity to a landfill site. Preference-based indicators add a further 'layer' to the physical indicators by eliciting some measure of preference for or against the physical risk. It is easy to see that the two generic indicators could differ substantially. Attitudes to risk are known to vary substantially and if they varied directly with income it is possible that any one socio-economic group might not be overly concerned about exposure to risks. Moreover, if more exposed groups felt they were compensated in some way, e.g. via lower property prices or closer proximity to employment opportunities, this may affect their preferences over risk[8].

As far as the physical indicators of risk are concerned, the literature works with several different concepts:

(a) emissions
(b) net emissions
(c) ambient concentrations
(d) exposure
(e) health risk

[8] The 'environmental justice' movement tends to disregard the preference-based approach since it sees equality of risk exposure as a 'right' that cannot be traded. Such 'lexical' orderings are the subject of a fairly extensive literature in environmental economics. In this case, however, the argument is further divided according to whether the advocates themselves have lexical orderings on behalf of those at risk, or whether those at risk have the lexical orderings. As several authors note, this makes distinguishing ideology from good research difficult.

The differences between these indicators are important and it is far from clear that the literature using emissions can be relied upon in terms of scientific integrity. The following discussion attempts to explain why.

Consider two areas R and P, with the R area occupied by high income ('rich') groups and the P area occupied by the low income ('poor') group. Then, on the emissions measure of EJ, an environmental injustice occurs if $M_P > M_R$, where M is emissions, e.g. tonnes of particulate matter. One hypothesis to explain this difference in emissions is that polluting industries locate in P rather than R in order to benefit from (a) lower wages and/or (b) lower political resistance to plant location. Both (a) and (b) are consistent with the polluting industries attempting to minimise costs. An hypothesis to explain the opposite result, i.e. $M_P < M_R$ would be that emissions are functionally related to income, a situation consistent with solid waste generation for example.

The net emissions and ambient concentration**,** A, measures of EJ can be considered together. Net emissions in any area are equal to actual emissions minus any 'exports' of emissions to other areas, plus any imports from other areas. The net emissions concept corresponds more closely to an indicator of damage or risk to health or ecosystems. For any emission source, it is necessary to have a dispersion model to establish what part of emissions affects a given population. Obviously, net emissions is a better indicator than emissions since any region could emit significant pollutants but not impose local risks of those emissions are exported. Similarly, an area importing pollution could face high risks even though it produces a low level of emissions. Ambient concentration is a better measure still, since it will tend to be related to damage.

Exposure represents a further refinement on the concentrations measure. Exposure differs from concentrations in allowing for the behaviour of the population at risk. For example, if low income groups work in outdoor occupations and high income groups work in indoor occupations, lower income groups will be more exposed to outdoor pollution even though ambient concentrations may be the same for both groups[9]. Similarly, the rich may be able to afford abatement measures such as double-glazing of windows to reduce exposure to noise. Access to medical advice and help may also vary with income groupings.

The final measure is total risk. This differs from exposure in allowing for personal characteristics of those exposed to risk, e.g. nutritional status, predisposition to ill-health, income-related behaviour such a smoking, etc. By and large, it can be hypothesised that the poor will, other things being equal, be more at risk than the rich.

Overall, the measurement of the state of the environmental risk matters for the EJ hypotheses. Ideally, a measure such as exposure should be used, but ambient concentrations are likely to be the closest measure obtainable. What is clear is that indicator such as emissions could be seriously biased.

Box 1 summarises the various measures of environmental risk.

[9] The importance of measures of exposure as opposed to concentration is stressed by Smith (1988). Note that the example given in the text could work the other way: indoor pollution may vary directly or inversely with income.

Box 1 Summary environmental indicators for use in distributional incidence studies
EMISSIONS For any given area occupied by a given income class, compute the emissions of various pollutants. Usually this applies to air pollutants but also has meaning for solid waste (waste generation rather than emissions). Emissions are a limited indicator due to (a) failure to account for imports and exports of emissions, (b) exposure, (c) personal characteristics that may magnify or reduce risk of harm. **CONCENTRATIONS** Net emissions (emissions - exports + imports) is an improvement on emissions, but concentrations, e.g. ppm or ug/m^3, offer a better indicator still. Estimating concentrations requires direct measurement of ambient quality or an estimate of emissions combined with a dispersion model. **EXPOSURE** Exposure to ambient concentrations allows for differences in activities among individuals, e.g. timing of work, location of work, and hence is a better indicator of risk than concentrations. However, exposure data are often limited in their availability. **RISK** Personal risk is a function of exposure plus personal characteristics such as predisposition to environmental insults (e.g. health state), plus other factors such as nutrition.

4.3 The spatial unit of account

A significant part of the EJ literature concerns itself with the right way to measure the geographical unit of account. It has been found that results of empirical studies are sensitive to the geographical scale of the study, ranging from small areas to large ones, and to the spatial resolution of the information, e.g. address codes, census tracts. In the US studies there is much debate about the use of SMAs (Standard Metropolitan Areas), census tracts, and zip code areas , these ranging from the largest to the smallest area. The smaller the area the less likely it is that the pollution variable will have meaning, while the larger the area the more are localised inequities likely to be overlooked.

As noted above, it is not always valid to equate emissions with exposure to risk, so that procedures to integrate the various dimensions of the environmental hazard with the various dimensions of the population at risk. The use of geographical information systems (GIS) is fairly recent but GIS has been applied to environmental equity issues - see, for example, Chakraborty (2001). In so far as GIS permits a better analysis of the data, more recent investigations using GIS are likely to be more reliable than the earlier studies which did not utilise the technique. As a general proposition, there are problems of assessing the validity of the various studies. One or two survey articles have made efforts to rank studies according to their scientific reliability, and choice of spatial unit figures prominently among the criticisms of various studies (e.g. Bowen, 2002). Accordingly, it is important not to treat the empirical literature as if each contribution has equal scientific status.

4.4 'Rights' based distribution versus preference-based distribution

The physical indicator-based notion of EJ tends to involve a moral judgement to the effect that a regressive distribution of environmental quality is unfair. The moral benchmark for this judgement is that an equal distribution of environmental quality is just, and, in turn, the foundation for this judgement is that all individuals have an equal 'right' to environmental quality. In some of the EJ literature this is further interpreted as requiring that all individuals should be exposed to 'zero' environmental risk, an empirical impossibility[10]. The weaker, and more realistic, form of the moral judgement is that (a) different income groups should be exposed to the same or similar non-zero level of risk, and (b) the risks should in some sense be 'acceptable'. All income classes might be exposed to the same level of risk but if that risk is unacceptable, then equity is breached. Similarly, risks to different groups might vary but both risk levels might be deemed acceptable. The notion of acceptability is not straightforward. What it usually means is that the environment should meet some standard of cleanliness set by law or public demand. This needs to be distinguished from a notion of acceptability whereby differential risks are significant, but the lower income group feels compensated by some other characteristic of their location (e.g. employment).

Rights-based activists would tend to argue that, while such a 'goals' may well be unrealistic, policy should nonetheless be aimed at moving towards them. However, there are several issues arising from the rights-based approach and these have all been raised in the EJ literature.

First, the distribution of any population is such that environmental quality per unit area will never be the same. People choose to locate where they do for many different reasons, so that adjusting environmental quality to be equal across all areas is very likely to be infeasible. Nonetheless, some environmental quality standards (e.g. air quality) are often set so that minimum quality levels are achieved, i.e. there is some notion of a threshold below which quality standards are deemed to be unacceptable.

Second, even if concentrations are equalised across areas, measures of exposure or total risk would probably not be equalised since these depend on human behaviour and prior characteristics.

Third, the rights-based approach tends to ignore costs. Equalising risks may well involve higher aggregate costs than if risks are differentiated. Equalising marginal costs of risk reduction would produce a minimum aggregate cost solution, but this would be consistent with risks varying

---

[10] The first law of thermodynamics is sufficient to show that zero risks are impossible, even if they can be thought of as desirable, which itself is a dubious judgement. See Wildavsky (1995).

location by location. While the rights-based approach would argue that risks and cost cannot be traded against each other, it remains the case that higher costs involve the sacrifice of other benefits that could be secured with the excess cost of risk equalisation. Those other benefits might also be the subject of an argument about 'rights', e.g. rights to health care or education. Equity in environmental risks could be at the expense of foregone rights across the population to non-environmental benefits.

Fourth, rights-based approaches assume *either* that individuals exposed to risk share the same notion of environmental justice, *or* that individuals' preferences should be over-ridden because individuals are unlikely to be well informed about the nature of risks. Both assumptions can be tested through survey techniques which elicit individuals' attitudes to the environment and to environmental risks[11]. As long as individuals' preferences do not coincide with the notion of equal risk, then the rights-based approach will have policy goals that are quite separate from those emerging from a preference-based approach.

Fifth, if preferences are deemed to be relevant, then one procedure for measuring them is to elicit willingness to pay. But since willingness to pay is likely to vary directly with income, high income groups will tend to have a higher willingness to pay than low income groups for environmental risk reduction. Rights-based advocates will therefore tend to dismiss the relevance of willingness to pay measures of risk preference. Their position on non-monetised preferences may vary from rejection of any preference-based approach through to seeking non-monetised expressions of preference.

Finally, contained within the willingness to pay approach is the idea that expressions of willingness to pay could well take account of associated compensation for tolerating higher risks, e.g. via lower property prices, employment opportunities, etc. This is the issue of self-selecting behaviour, i.e. what is observed may be a set of individuals who have chosen to locate in low quality areas because they have traded off the associated costs with other benefits (Hite, 2000). This issue is complicated by the sequence of events that generate an environmental risk. For example, poor people living in an area that is developed for an airport may find themselves to be losers if property prices fall due to airport noise, pollution and congestion. But they could secure windfall financial gains if the airport attracts employment and the demand for housing rises. Those moving into the area after the airport is built could be compensated for any environmental problem through house prices, employment opportunities etc. Higher income groups might move out of the affected area, suffering a loss if property prices are depressed by the airport and experiencing a windfall gain if property prices rise[12]. Thus what matters is the net benefit or cost rather than just the cost of any risk. Adoption of a monetised preference approach to environmental equity can therefore quickly produce wholly ambiguous results, with a final judgement resting on detailed analysis of the welfare gains and losses for each group of individuals affected by the risk-creating activity.

At the heart of the different approaches to EJ is the familiar debate **over economic efficiency versus equity**. As will be shown shortly, iniquitous outcomes may well be economically efficient. The EJ movement selects equity as the relevant goal, usually completely disregarding

[11] This argument holds independently of how preferences are measured, e.g. there is no presumption that preferences are measured by willingness to pay to avoid risks.
[12] These arguments in the context of airport noise are presented in some detail in Walters (1975).

efficiency[13]. The preference-based approach tends to assume efficiency is a 'higher' goal, the usual argument being that the creation of maximum social surpluses permits some of that surplus to be used to correct inequity. Somewhat surprisingly, little appears to be known about how individuals 'trade-off' equity and efficiency. Atkinson et al. (2000) have shown how survey-based approaches to trade-offs between apparently competing principles of financial burden sharing for environmental programmes can produce robust indicators of individual preferences. It remains to be seen how people trade-off notions of equity against changes in total net benefits from such programmes. It should also be noted that the rights-based approach is consistent with inequity being the result of economic forces that produce an efficient but inequitable outcome, and it is also consistent with inequity being a deliberate outcome of some exploitative process, including racism. The preference-based approach tends to assume away notions of exploitation of the poor or minority populations.

5 WHY DOES ENVIRONMENTAL INEQUITY ARISE?[14]

This section describes alternative (although sometimes overlapping) explanations for the rise of environmental inequity.

5.1 Income inequality and Tiebout local public goods

The simple fact of income inequality is sufficient to produce differential environmental conditions faced by 'rich' and 'poor', provided environmental quality varies with spatial location. Since willingness to pay for environmental quality is itself a function of income[15] then, regardless of the cost of supplying that quality, the rich will receive a higher level of quality than the poor. Figure 4 illustrates this simple proposition.

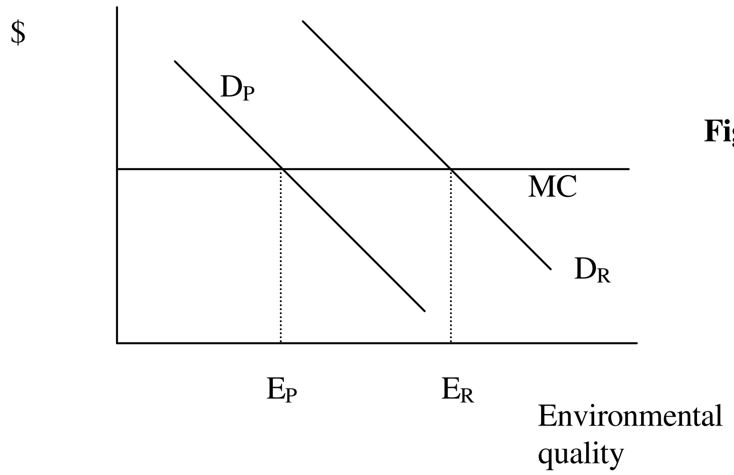

Figure 4

[13] We take efficiency to mean the maximisation of net social benefits or the minimisation of net social costs, where benefit and cost derive their meaning from individuals' preferences as measured by willingness to pay or willingness to accept.
[14] In discussing the various modes of causation we do not consider the notion, conveyed in significant parts of the EJ literature, that inequity is the result of racism. Racial prejudice will produce inequity between income groups, the focus of this report, if minority groups are correlated with low incomes, as tends to be the case. Omitting a discussion of racism as a causal factor is not intended to downplay its potential importance. For a discussion see Pellow et al. (2001).
[15] We investigate just what this relationship is in Section 8.

(D = demand, R = rich, P = poor, E = environmental quality, MC = marginal cost of supply).

Implicit in this explanation is that individuals move location to equilibrate their demand for environmental quality with its cost of supply. The reason for this assumption is that environmental quality tends to have public good characteristics[16], so that what is supplied to one group is automatically supplied to the other group within the jurisdiction of the public good. However, environmental quality can vary from location to location, so that, within any location space, environmental quality is a local public good, but environmental quality is not a pure public good across all locations. The notion that individuals with varying demands for local environmental quality will spatially allocate themselves in this way is due to Tiebout (1956). Various EJ models stress either the 'push' factors, i.e. for the rich, the presence of risk-creating activities as deterrents to residence, or the 'pull' factors, i.e. the relative attractiveness of areas without risk-creating activities. Similar 'pull' factors will operate for the poor, since high risk areas may be areas with greater employment opportunities (Liu, 2001), tax revenue needs, or lower priced property (Hite, 2000). In this respect the models are not different to the basic Tiebout model above. However, non-economic models based on these push-pull factors tend to emphasise the feedback effects as well. As low income groups migrate to high risk areas in search of the associated benefits, so high income groups are further deterred from staying since they have preferences for locating in more homogeneous high income areas (Been, 1994).

In what sense does an environmental equity problem arise with the Tiebout model? On the rights-based approach, rich and poor would have an equal entitlement to an acceptable level of environmental quality, and E_P in Figure 4 might be outside the range of 'acceptability'. On the preference-based approach, the difference between E_P and E_R would have a justification in the fact that both rich and poor have freely chosen their equilibrium, within the constraints set by their incomes. Indeed, it is possible that environmental quality differences are capitalised in some other good, such as property – see, for example, Brainard et al.(2003b). If so, the poor would, other things being equal, face lower property prices than the rich, exactly compensating them for the lower environmental quality. Note also that, had the two groups the same income, and the same tastes, then their demand curves for environmental quality would be coincident, and each would demand the same level of environmental quality. What some rights-based analysts are drawing attention to, therefore, is not so much the injustice of differing environmental quality levels, but the apparent injustice of different income levels: i.e. it is the distribution of income they are objecting to. The contrasting outcomes of the two approaches amply illustrate the difficulty in defining what 'environmental justice' means. On the rights based approach, an injustice remains. On the preference-based approach, there is no environmental injustice. The poor have the amount of the good they desire, as have the rich. As Been (1994) puts it:

> 'As long as the market allows the existing distribution of wealth to allocate goods and services, it would be surprising indeed, if, over the long run, LULU [locally undesirable land uses] did not impose a disproportionate burden upon the poor'.

[16] We take a public good to be one that, if provided to R would also be provided to P, without R being able to prevent P from securing the good. These are the attributes of joint consumption (non-rivalry) and non-exclusion.

Evidence that local public good disamenities are compensated for by differences in wages and house prices is provided in Blomquist et al. (1988). This study estimates a 'quality of life index' for urban areas of the USA and concludes that:

'..compensation for location-specific, non-traded amenities takes place in both the labor and housing market and that the amount is substantial' (p105).

5.2 Income inequality and pure public goods

The Tiebout hypothesis cannot hold if the public good is 'global', i.e. if the publicness extends across all feasible locations. Then, movement would not secure any change in environmental quality. Baumol (1972, 1974) and Baumol and Oates (1988) produce a theory of why *a form* of environmental injustice (as it would now be called) would come about in the case where environmental quality is a pure public good. Their approach is illustrated in Figure 5. Rich (R) and poor (P) have budget lines RR' and PP' respectively. These are drawn parallel to indicate that R and P face the *same price* of the public good E, environmental quality. Given their respective indifference curves, as shown, P would demand E_P of environmental quality, and R would demand E_R. Their respective equilibria are shown as C and A. Note that the similarity of the shapes of the indifference curves assumes that preferences are similar across the two income groups. However, since environmental quality in this case is a pure public good, whatever is supplied is supplied to both groups. What is supplied is therefore some compromise achieved by the political system. As shown in Figure 5, the quantity supplied, E, is roughly the average of the two demands, but, clearly, it could be closer to E_R or to E_P, depending on the relative political power of the two groups. But if E is supplied, then both rich and poor move to lower indifference curves than if each could have had the quantity they demanded (the Tiebout case). Their respective positions are given by D and B. The poor are now 'oversupplied' with environmental quality (D>C) and the rich are 'under supplied' (B<A).

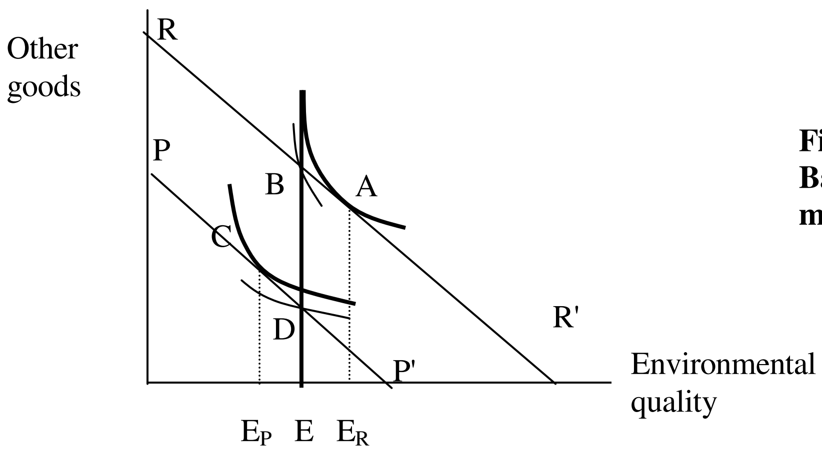

Figure 5: the Baumol-Oates model

In what sense does the Baumol-Oates solution produce environmental inequity? In terms of the equity notions so far introduced, it does *not* produce environmental inequity. This is because both rich and poor receive the *same* level of environmental quality, E. The concept of equity here rests on the view that it is more unfair to make the poor pay for something they do not want (E_PE of environmental quality) than to fail to supply what the rich want (by the margin EE_R)[17]. Moreover, this inequity is larger the closer E is to E_R, a result that might be expected if the rich are politically more powerful than the poor. In terms of Figure 5, P's willingness to pay for the 'excess' amount of the public good they receive is lower than the price they are being charged for it, while R's willingness to pay for the amount they fail to secure is greater than the ruling price. Expressed as a proportion of income, therefore, the poor are likely to have a larger welfare 'deficit' than the rich.

Once again, the analysis shows how difficult it is to define environmental equity: even if the rich and poor have access to the same level of environmental quality, a form of inequity can arise in so far as there are unmet preferences for the rich and 'forced' oversupply to the poor. Environmental benefits will tend to be distributed in a 'pro-rich' fashion.

The Baumol-Oates result is the outcome of market forces and the political system. Market forces alone, reflecting the prevailing distribution of income, can produce the result. But it is more likely that the unbalanced exercise of political power will bias the result further against the poor. To this end, the Baumol-Oates model combines market and political forces, the latter reflecting a degree of 'exploitation' of the poor by the rich.

Certain assumptions underlie the Baumol-Oates result. First, the preferences of R and P for the environment are assumed to be similarly structured. If the poor had strong preferences for the environment relative to those of the rich, then it is easy to generate the result that the poor get the right amount of environmental quality and the rich too much[18]. A further problem is that the analysis assumes rich and poor face the same price for the public good. But methods of financing the good may have progressive structure, e.g. income tax, whereby the rich face higher average taxes than the poor. Strictly, what matters is the marginal rate of taxation and this too could differ, being higher for the rich. If so, the budget lines in Figure 5 are no longer parallel, and it is possible to secure the opposite result to that shown in Figure 5[19]. Overall, however, the Baumol-Oates model provides an explanation for inequity in the specific sense of under and over-supply of the public good.

5.3 The spatial shifting of externalities: political power and inequity

It is widely argued that the Coase theorem (Coase, 1960) can be adapted to suggest that any risk-creating activity, say a factory or waste disposal site, will be located where environmental externalities are minimised (e.g. Hamilton, 1993). The intuition is that the risky activity will be located in low income areas because the willingness to pay to avoid the facility will be lower than in a high income area. Hence firms with a choice of locations will choose that site where, if compensation had to be paid, the sums paid out would be the lowest. In fact, compensation pay-

[17] In Figure 5, the poor's willingness to pay is given by the slope of their indifference curve at D, and this is less than the price of the public good, shown by the slope of the budget line. The rich's willingness to pay exceeds the price at B.

[18] Such a case would be illustrated by making the indifference curve of the poor very steep and that of the rich very shallow.

[19] Baumol and Oates (1988) attach little importance to this case but this may reflect the nature of marginal taxation in the USA. Marginal tax rates do vary significantly in other countries. Note that the compromise supply of the public good is now assumed to be the outcome of a political bargain based on post-tax bargaining functions rather than pre-tax ones.

outs would be only one of a class of costs to the firm that would be minimised by siting in low income areas. The model can be developed further to include political activity, the presumption being that low income groups will not organise collectively in as efficient a way to oppose the risk activity, whereas high income groups will. Thus, even without the notion of willingness to pay, a model of political collective action is sufficient to generate an efficient outcome - the externality will be minimised (Becker, 1983). These outcomes are illustrated in Figure 6. Figure 6 shows environmental damage along the horizontal axis and money on the vertical access. The downward sloping line is the marginal profit curve (MΠ) of the polluting activity, such that the firm will maximise profits when MΠ = 0. The upward sloping curves are the marginal environmental damage (MD) curves in rich and poor locations, MD_R lying above MD_P due to income differences. Beginning with the assumption that the polluter has the property rights, the Coase theorem tells us that the two parties (rich or poor, and the firm) will bargain to achieve an optimal amount of externality. If there is only one possible location, this optimum is either A (if the polluter locates in the rich area) or B in Figure 6 (if the polluter locates in the poor area).

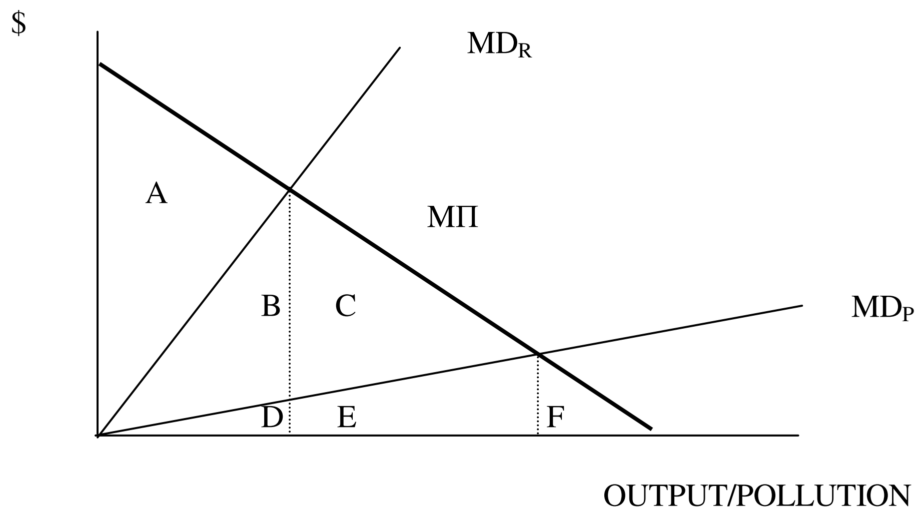

Figure 6: Coaseian Bargains and Environmental Equity

Suppose the property rights rest with the sufferer. Then, the polluter must pay compensation to the sufferers and the starting point is the origin in Figure 6. It will pay the polluter to offer compensation up to the intersection of MD_R and MΠ in the rich location and the intersection of MD_P and MΠ in the poor location. Assuming compensation just offsets damage, the polluter's net profits will be A+B+D – (B+D) = A if he locates in the rich location, but A+B+C+D+E-(D+E) =A+B+C if he locates in the poor area. Hence, from the polluter's point of view he will prefer to locate in the poor area. This is consistent with the political model which says that poor locations will be preferred by polluters..

From an environmental point of view, there is more *physical pollution* if the polluter locates in the poor area. Hence, if pollution is a public 'bad', each individual in the poor area is exposed to more pollution than would be the case if the polluter located in the rich area – there is an environmental justice problem. However, the *money value of environmental damage* (the area under the MD curves) could be less or more in the poor area than in the rich area (compare B+D with D+E: area D is common but we do not know if B is less than or greater than E). Hence, on the preference-based approach we cannot say if there is an environmental justice problem. In so far as those preferences show up in willingness to pay, and given that willingness to pay is conditioned by income, the inequity is really one of unequal incomes. Finally, from the standpoint of *net social benefits* locating in the poor area is to be preferred to locating in the rich area since (A+B+C)>A, illustrating the point that what is efficient is not necessarily the same as what is environmentally just.

Now suppose, once again, that the polluter can choose to locate in the rich or poor area but this time the polluter has the property rights. The starting point is now $M\Pi=0$ and, if bargaining occurs, the same two equilibria will emerge, one for the rich location, one for the poor location. But, so long as the sufferer pays the polluter just enough to offset the lost profits from reducing his level of activity, the polluter is no better off in the poor area than in the rich area. In the poor area, he moves to the intersection of $M\Pi$ and MD_P but he receives payment equal to area F from the sufferers. His total net revenues are therefore the same as he gets when $M\Pi=0$. This is obvious since he will not enter a bargain unless he is at least no worse off. The same argument holds for locating in the rich area: he moves to $M\Pi = MD_R$ where his profits are A+B+D, but receives compensation of C+E+F from the rich sufferers. He is therefore no worse off. From the point of view of *physical pollution* choosing the poor area is worse and, once again, from the standpoint of monetary value of damage we cannot say which is better. Hence, if the polluter has the property rights, he should be indifferent to where he locates.

It is in fact quite possible to argue that the polluter could deliberately locate in the rich area. He may do this if he has the property rights and believes that the rich will *over-compensate* him for output restrictions more than the poor will over-compensate him. This is akin to rent-seeking activity, the area under MEC_R being larger than the area under MEC_P.

Hamilton (1993, 1995) notes an important modification to the Coase-Becker argument above. This suggests that the regressive nature of risk-generating activity will be further aggravated by the lower likelihood of collective action by the poor. But, in contrast to the Coase-Becker model, this further reinforcement of the bias in siting decisions may be inconsistent with the result that the bias is optimal from an efficiency point of view. Essentially, the locations with the least political opposition may not be the locations with lowest willingness to pay to avoid the externality if willingness to pay is not truly 'revealed' in political activity. Whereas the Coase-Becker result assumes that both R and P organise themselves efficiently, Hamilton focuses on the probable differences in the ability of individuals to organise themselves. Even if willingness to pay to avoid the offending activity was the same in R and P, these differences in ability will show up as lower *expressed* political opposition[20]. A 'wedge' is driven between the (welfare) losses in P and the 'voice' expressing that welfare loss. Hamilton (1993) tests his model by observing that the siting of hazardous waste facilities in the USA between 1987 and 1992 is correlated with low voter turnout in elections. This result may not be consistent with the Coase-Becker outcome that overall social costs are minimised (or net benefits maximised, as in Figure 6). This is because the siting decision depends on expected externality costs multiplied by the

[20] *Why* these differences in collective organisation exist is yet another major question of interest, but not one pursued here.

probability that residents will effectively oppose the siting decision. Social costs could be higher in poor districts but, since the probability of translating those welfare losses into litigation, lobbying etc. is small, the effective cost to the firm is less. Hamilton (1999) shows that TRI plants with the highest toxic releases reduced their emissions more. As voter turnout increased, so emissions declined. Other studies, e.g. Arora and Cason (1999), also find that pollution is inversely correlated with proxies for political action: the stronger the voter turnout or the proxy for voter turnout, the less the emission level. In the USA this effect tends to work via the 'right to know' legislation concerning toxic releases.

It is possible to construct various models that produce similar results to the Coase-Becker outcome and the Hamilton outcome. For example, the rich are more likely to be able to move location than the poor. The rich are likely to be more mobile occupationally. Even if they fail to prevent a polluting activity being located in their area, they are more likely to be able to move away from the area once the siting has occurred. If so, the differences in mobility will act just like the differences in political activity - the poor will reside in more polluted areas than the rich. A similar result emerges from models of behaviour when different income groups face a given environmental situation. For example, suppose the *existing* level of environmental quality is the same in a rich and poor area. Then the hypothesis is that the rich will engage in clean-up activities more than will the poor. So long as rich and poor occupy different locations, then an environmental inequity arises because the poor will then be exposed to more risk due to their inaction, while the rich will have cleaner environments[21]. However, if rich and poor occupy an area in which clean-up activities have public good characteristics, then the poor will benefit as free-riders from the activities of the rich. Such outcomes are consistent with everyone behaving in a self-interested manner, but there is some evidence to suggest that richer people may also behave more altruistically in this context, in line with a prediction of Olson (1965) - see, for example, Cardenas et al.(2002).

5.4 Environmental discrimination

Differences in political power also occupy a central role in most political explanations of environmental inequity. These models tend to be qualitative and based on case studies of individual siting decisions and the various reactions to them by different groups of stakeholders. Political power is unevenly distributed across stakeholders, with minorities, low-income and immigrant communities having the least power. Inequity is then the outcome of a struggle for resources, where resources include clean environments, recreational facilities and the work environment (Pellow et al. 2001). Such struggles result from competing self-interests but also from racism and discriminatory attitudes among individuals and institutions. Testing hypotheses about discrimination is obviously complex and controversial. Much of the evidence in favour of the discrimination hypothesis is based on the *outcomes* of the relative burdens of environmental and other damages, i.e. the fact of inequity is seen as evidence of discrimination. But the obvious problem is one of demonstrating discriminatory *intent*, something that is not easy to do in any quantitative manner. This perhaps explains the heavy reliance placed by environmental justice advocates on narrative case studies.

[21] There are obvious 'moral' issues here. If the poor *choose* not to opt for clean-up and the rich do, the preference-based approach would conclude that the outcomes are optimal. The EJ movement, however, would argue that the inability or unwillingness to organise is itself a function of the initial income inequality and hence the poor should be helped in some way to organise themselves. Indeed, much of the EJ movement is concerned with exactly this process.

5.5 'Ecological' explanations for inequity

Ecological models tend to stress the dynamics of land use change and social grouping that takes place independently of any siting of risk-creating activities. Models are akin to invasion-succession sequences in standard ecology: incoming populations gradually expand and displace previous populations. If high income groups believe that low income groups lower property values and the social value of the community generally, then the higher mobility of high income groups will lead them to move out of a given area. If low income or ethnic minorities come into the area, the process of 'ghettoisation' might expand into surrounding areas. The presence of a risk-generating activity might then be accidental: what is observed is a post-siting decline in the average income level which then misleadingly appears to be associated with the activity (Liu, 1997). Put another way, the statistical association between low income areas and higher risk activity, if it exists, tends to reflect baseline population dynamics rather than a causal process. Again, such models are not readily tested for in a quantitative fashion and resort is usually made to historical case studies to illustrate the sequence of events.

5.6 Conclusions on explanatory models of environmental inequity

While it is a simplification, the following proposition helps to 'fix' ideas about the competing explanations for environmental inequity. Where the environmental good in question is a pure public good, the spatial nature of which traverses all relevant areas, there is no real possibility that environmental inequity, as defined by the EJ literature, will exist. Essentially, all income groups will be exposed to the same level of risk, ignoring any personal characteristics that make one group more disposed to the risk than other groups. That level of risk may be one associated with the economically optimal provision of the public good or the 'political' optimum brought about by competing demands for the good. However, as the Baumol-Oates public good model shows, there is a form of inequity in so far as any compromise supply of the good results in over-provision to the poor and under-provision to the rich. It is important to note, however, that the Baumol-Oates notion of inequity is not that adopted by the EJ literature.

In practice, few environmental goods are pure public goods. A great many of them take on the features of local public goods, so that the consumption of those goods varies by location. Tiebout-type models emphasise the fact that differences in income, and hence in willingness to pay, will set up migratory processes that will lead low incomes to consume a lower quality environment than the high income groups. Those non-economic models that stress push and pull factors, tend to fit into this model as well, although they are arguably richer in that they stress the cumulative effects of changing population characteristics on the migration process.

Political explanations for any low income - high risk association can also be fitted into the general Coase-Becker framework which stresses the role played by anticipated costs to a firm locating in a given area. Such costs -e.g. litigation, transactions costs - are likely to be higher in high income areas where the ability to organise and lobby against a siting decision is higher. As shown, independently of the allocation of property rights, the Coaseian process produces outcomes which will tend to site the offending facility in a low income area. The Coase-Becker model is both a 'positive' explanation of why this outcome occurs, but it also has normative content in suggesting that the result is, in any event, optimal from an economic efficiency standpoint. As noted earlier, what is efficient may not be equitable, and there is no 'meta principle' to determine whether equity is more or less important than efficiency. Understandably, therefore, the EJ literature is sharply divided on the importance of efficiency. Hamilton's (1993) work does sound a caution about Coaseian outcomes being efficient if true willingness to pay by low income groups diverges from expressed political opposition.

What might be called the 'market dynamics' approach to equity therefore acknowledges that outcomes may be inequitable but that the inequity is a direct result of inequality of incomes. If the prevailing distribution of income is itself optimal, then, as the quotation from Been (1994) suggests, there will be unequal exposure to risk. What much of the EJ literature is therefore contesting is the optimality of the prevailing income distribution. Unequal risks are simply the outcome of unequal incomes.

Finally, care has to be taken to distinguish positive and normative issues. Much of the literature tries to explain why inequity arises, and this is an exercise in positive analysis. However, these analyses are nonetheless driven by the implicit or explicit judgement that unequal exposure to risks is unfair, and that is a normative judgement.

6 THE EMPIRICAL EVIDENCE ON ENVIRONMENTAL INEQUITY: PHYSICAL MEASURES

The extensive literature on physical measures of environmental inequity is primarily American. Caution therefore needs to be exercised in supposing that, even if the American literature has a consensus finding, the finding can be generalised to other OECD countries.

The various studies use different approaches, resulting in different indicators of inequity. Studies focusing on landfill/hazardous waste sites tend to take a socio-economic indicator, say income per head or per household and compare it to 'with' and 'without' site locations. If, for example, locations with sites are systematically associated with lower incomes than locations without sites, then this would be regarded as evidence of environmental inequity. The degree of risk is typically not measured, i.e. the risk takes a [0,1] measure: either there is a site or there is not. Results may then be formulated as 'Poorer people are X times more likely to be located near to a risky site than are rich people', where 'poor' and 'rich' themselves need to be defined. Air quality studies, on the other hand, have continuous data that can be compared to socio-economic information. Results are presented in various forms. Areas may be classified as 'low' air quality and 'high' air quality based on the continuous data, reducing the location characteristic to a [0,1] form again. More elaborate use of continuous data often involves regressions of the form:

$$Poll = aA + bB + \ldots + rR + yY$$

where Poll is pollution, a..y are coefficients, A,B are explanatory factors other than race (R) and income (Y). If y is negative and statistically significant, there would be evidence of income inequity. Many studies simply report correlations, i.e. ..a value of r or r^2, between pollution and income (race). A few studies report indicators such as 'poor persons per square kilometre' and correlate this with pollution. An indicator of the form 'pollution per unit income' appears not to be reported in any study, but a 'pollution elasticity' is reported by Kahn (1998) and Khanna (2001). Kahn finds an inverted 'U' curve (i.e. an EKC - see Section 4.1) for vehicle emissions in California. As median household income increases so pollution at first increases and then decreases. Khanna finds the opposite result for the same data, i.e. a 'U' shaped curve linking pollution and income per head.

Section 5 suggested some plausible combinations of market and political forces that could give rise to environmental equity, where inequity is defined as in terms of the poor being exposed to greater environmental risks than the rich. This section reviews the empirical literature. In so doing, we make the assumption that, unless specific errors in the studies have been pointed out, all studies are equally valid. This is a strong but unavoidable assumption. Most of the empirical literature emanates from the USA where, as noted earlier, the environmental justice movement has a strong foothold in political discourse and in legislation. Limited information appears to exist for other countries.

Europe

Table 1 shows the evidence for Europe. While the evidence is very limited, the data for the UK suggest that the *existing* distribution of risks is biased towards the poor. For the spatially wider public goods, such as air pollution control, policy to reduce those risks would therefore be 'pro-poor'. For locally confined risks, e.g. from waste sites, the outcome would depend on the specific targeting of policy towards the areas of risk. It should be noted however that several of the UK studies use *emissions* as the relevant pollution indicator, imparting an unknown degree of error to the results.

Table 1 **Social incidence of environmental damages in Europe**

Study	Pollutant or hazard	Finding
McCleod et al. 2000. England/Wales Uses GIS.	SOx, NOx, PM_{10} *concentrations*	Pollution negatively associated with lower social class index, i.e. inequity exists, but rich in SE England exposed to higher pollution than poor in other regions.
Pye et al. 2001, UK Uses GIS	NO_2, PM_{10} *concentrations*	Weak positive correlation between pollution and social deprivation. Clean air policy simulations benefit the poor most.
Friends of the Earth, 2001, UK	Carcinogenic factory air *emissions*	Highest emissions occur in most socially deprived areas.
Walker et al. 2000. UK	Hazardous substances - accident risk	Risk correlated with ethnic minority population concentrations
Stevenson et al. 1988. UK	Road traffic emissions	Respiratory illness correlated with emissions and low income
Bateman et al. 2002. Birmingham, UK	Air pollution	Pollution correlated with ethnicity even when income controlled. Income relationship regressive.

Bateman et al. 2003; Brainard et al. 2003a: Birmingham, UK	Noise	Noise levels and socio-economic deprivation very weakly correlated. Very weak association noise and ethnicity. Night-time noise correlated with deprivation.
Brainard et al. 2003b. Birmingham, UK	Green space	Income relationship regressive
Kruize and Bouwman, 2003. Netherlands	Noise, air pollution, green space, safety risks, waste facilities	All risks regressively distributed other than aircraft noise.

6.1 North America

Tables 2 and 3a/3b look at the far more substantive North American evidence[22]. Table 2 deals with the early studies and Table 3 with more recent studies. A review of early studies is provided in Cutter (1995) and of early and late studies in Bowen (2002). Hamilton (2003) provides an extensive review that focuses on hazwaste. The very early studies all related to air pollution, other than the Berry (1977) volume which covered many forms of pollutant. The general finding was that damage was higher the lower the income level, but with qualifications shown in Table 2. In the 1980s focus shifted to hazardous waste sites, and in the 1990s (Table 3) coverage included toxic releases, waste sites and air pollution, particular targets of the environmental justice movement. It is important to note that several of the early politically influential studies (e.g. UCC, 1987; GAO 1983) have been severely criticised[23].

Table 2 Social incidence of environmental damages in the USA: early studies

Study	Pollutant or hazard	Finding
Freeman 1972	TSP, SOx in 3 cities. *Concentrations*	Low income groups have higher pollution within cities, but relationship breaks down across cities
Zupan, 1973. New York city New York Met area	SO_x, TSP *concentrations* CO, SO_X, TSP *emissions*	Pollution correlated with low incomes
Harrison 1975	CO, NO_X, O_X. *Concentrations*. Benefits of auto-emissions control policy	Pollution reductions unrelated to income in metropolitan areas. But pro-poor benefits in

[22] Goldman (1994) has pointed to the US bias in EJ studies, calling for similar studies in other countries.
[23] Table 3 does not pretend to be comprehensive as there are many studies of the kind listed, varying in sophistication. Studies have been chosen according to a rough check on the number of citations.

		urban areas.
Berry 1977	PM10, SOx Chicago *Concentrations*. PM10, SOx 12 cities *Concentrations*. Noise, solid waste	Low and middle income groups bear higher pollution Low income groups bear higher pollution Unclear
Asch and Seneca 1978	PM10, NO2, SOX. *Concentrations*. 23 states	(Generally) low income groups bear higher pollution burden
Harrison and Rubinfeld 1978	Air pollution, Boston	Air pollution improvements pro-poor
Bullard, 1983	Hazwaste sites, Houston	Black residents more exposed to sites. Study severely criticised by Been, 1994 and Bowen, 2002.
US GAO, 1983	4 hazwaste sites in SE states	Sites correlated with Black populations and poverty. Criticised by Bowen 2002.
UCC, 1987	Hazwaste, national level	Sites correlated with Black and minority populations. Severely criticised by Bowen 2002.
Rose et.al. 1989	Environmental damage from surface mining, Virginia	Damage distributions compared to income gains from mining. Lower income households bear greatest net losses

Table 3a Social incidence of environmental damages in the USA: recent studies

Study	**Pollutant or Hazard**	**Finding**
Brajer and Hall 1992, California	O_3 and PM_{10}	Pollution correlated with low incomes, Black and Hispanic communities. Criticised as 'poor' research by Bowen 2002.

Mohai and Bryant, 1992	Hazwaste sites in Detroit	Distance to site regressed on race and income. Inequality confirmed. Severely criticised by Bowen 2002.
Zimmerman, 1993	Hazwaste sites, national	Hispanics and Blacks correlated with sites but no link to income
Burke, 1993	TRI releases. *Emissions*. Los Angeles County	Minority status and low incomes correlated with emissions
Anderson et al. 1994 Anderton et al. 1994a Anderton et al. 1994b	TSDF sites **. National.	1980 sites not inequitably distributed by race/ ethnicity but some evidence of income inequality. 1990 sites more inequality by income. There is 'almost no support for the general claim of environmental inequity'. Goldman and Fitton (1994) find a race link if zip codes are used.
Perlin et al. 1995	TRI releases*. *Emissions*	Emissions positively associated with *higher* incomes, but also with ethnic minority presence.
Gluckman, T and Hersh, R. 1995	Toxicity-weighted TRI releases, and storage of hazardous substances, Allegheny County, Pittsburgh	Proximity to hazards mainly affects low income groups. Mortality risks far less clearly related. Warn against generalisations.
Bowen et al. 1995	TRI releases. Emissions. Cuyahoga County (includes Cleveland)	Reverse link race and emissions at census tract level but positive link at county level
Kriesel et al. 1996.	TRI releases. *Emissions*	Results depend on specification of model, e.g. race and income inequality exists when only they are included in model, but do not exist when other variables added.

Yandle and Burton 1996	Hazwaste sites, Texas	Correlation with low income White areas
Cutter et al. 1996	TSDFs, S.Carolina	No association at census level, some indication of sites being in higher income White areas otherwise
Boer et al. 1997	TSDFs. Los Angeles County	Minorities linked to sites but not income
Brooks and Sethi, 1997	Toxicity weighted TRI releases. *Emissions.*	Low income, low education, minority populations correlated with pollution. These groups have benefited most from improvements.
Bae, 1997. Los Angeles	O_3, CO, NO_2, SO_2, PM_{10}, Pb *Concentration* exceedances above standards	Existing risks borne more by poor, therefore benefits of policy pro-poor
Been and Gupta, 1997	Hazwaste sites	New sites 1970-1994 were *not* located in African-American areas. But evidence of Hispanic bias. No correlation with poverty.
Hockman and Morris, 1998	Hazwaste sites and incinerators, Michigan	Race correlated with sites, especially incinerators
Hamilton and Viscusi, 1999	Hazwaste: Superfund priority list	Non-white populations disproportionately represented near sites. Income regressively related.
Arora and Cason, 1999	TRI. *Emissions* US-wide, zip code level	Non-whites, low incomes and unemployment correlated with TRI especially in SE areas, and especially in non-urban areas. Demographic variables proxy for political action, inversely correlated with TRI
Hite 2000	Landfills in Franklin County, Ohio	Evidence of racial inequity but no evidence of income inequity. See text for discussion.

Millimet and Slottje, 2000	TRI. *Emissions*.	Uses Gini measure of pollution inequality across and within states. 1988-97 substantial inequality exists but stable over time at both levels. 1998 saw rise in both levels of inequality. Reduced air pollution *increases* inequality.
Kahn, 2001. California	CO, NO_2, O_3, SO_2, PM_{10} *Concentrations*	Apart from O_3, poor experience higher pollution than rich. Poor have experienced a relatively greater improvement in air quality 1980-1998
Haynes et al. 2001 Cuyahoga, Cleveland	TRI releases. *Emissions*	Higher releases correlated with Hispanic population, and low housing values, but doubts about findings.
Chakraborty 2001	Accidental releases of extremely hazardous substances, Hillsborough County, Florida	Non-white population and poverty linked to number of releases
Atlas, 2001	TSDFs, national.	No pattern of TSDFs or their risks being inequitably concentrated in disproportionately minority or low income areas

Note: TRI = Toxic Releases Inventory. TSDF = Transfer, storage and disposal facilities

Table 3b Social incidence of environmental damages in Canada

Study	**Pollutant or hazard**	**Finding**
Handy, 1977, Hamilton	Air pollution: dust and sulphates	Income negatively associated with pollution, i.e. inequity
Jerrett et al. 1997 Ontario	National Pollution Release Inventory	Income *positively* correlated with pollution, i.e. no inequity
Harrison and Antweiler, 2002	National Pollution Release Inventory	No association between releases, transfers and income

Use of the Toxic Release Inventory is interesting because of the way different authors have treated the data. Some studies simply aggregate pollutants in the TRI regardless of their difference in toxicity (Perlin et al. 1995; Millimet and Slottje, 2000), and this could be inconsistent with what should be risk-based exposure indicators. Some authors seek to weight the individual releases by some measure of toxicity (Brooks and Sethi, 1997; Arora and Cason, 1999). The extent of bias from using unweighted data is not easy to gauge, although Millimet and Slottje (2000) argue that weighting provides little value-added since the chemicals probably do not vary much by toxicity. The TRI is also an emissions inventory rather than an exposure inventory, whether toxicity is allowed for or not, and it was noted earlier that this is a potential distortion of the exposure-impact relationship..

The Millimet-Slottje analysis requires discussion because it utilises an imaginative approach to policy implications. First, they find substantial inequity in the prevailing distribution of pollution. Second, they further decompose pollution into releases to land, water and air. For 1997 (1998 was an unusual year, see Table 3). Air emissions accounted for over half of the 'inequality' of social burdens, land for about 30% and the remainder by water and underground releases. An interesting finding is that policy designed to reduce land and underground emissions is *inequality reducing* at both county and state level, but that comparable policy on air emissions *increases inequality*. This finding contrasts with much of the EJ literature. Third, the analysis also estimates an 'environmental welfare function' in order to investigate the trade-off between pollution and inequality (pollution reductions being treated as increases in income)[24]. The aim of this function is to explore the trade offs in contexts where, as found for air pollution, pollution declines but inequality rises. Clearly, any such trade-off has to incorporate some measure of inequality-aversion and the model is tested for varying levels of aversion. Ignoring 1998, the level of 'environmental welfare' increases over time, even when fairly high degrees of inequality aversion are incorporated[25]. Finally, Millimet-Slottje produce some partial tests for the 'compensation hypothesis' introduced in Section 3. In their case they investigate whether wages vary with pollution, so that damages from pollution may be offset (at least partially) by higher wages. They do find evidence for this effect. The Millimet-Slottje study is considerably more sophisticated than the vast majority of contributions to the EJ literature since (a) it shows that prevailing inequality in pollution incidence does not necessarily translate into the finding that reducing pollution will reduce inequality; (b) it makes an explicit attempt to investigate the trade off between reduced pollution and increased inequality, and (c) it provides some evidence for the compensation hypothesis.

The other study that adopts a strikingly more realistic approach to the environmental justice issue is Hite (2000). The reason for this is that the results emanate from a random utility model in which individuals choose location characteristics so as to maximise personal ‘utility’. As noted above, simply observing the location of risky installations and correlating that location with population characteristics fails to account for any trade-offs that individuals may make between the risk and other characteristics of the area. The differences in approach tend to produce a divide in the empirical literature. Whilst perhaps a simplification, 'rights based'

[24] The 'environmental welfare function' is given by $EW = -Y^*(1+G)$ where Y^* is mean income and G is the (income) Gini coefficient.

[25] On measures of inequality aversion, see Section 9

authors favour the statistical association measures, and 'preference based' authors favour the trade-off or compensation approach.

6.3 Conclusions on environmental equity using 'physical' indicators

Summarising the findings of the literature in Tables 1-3 is difficult. First, studies vary widely in their modelling sophistication. Second, choice of spatial unit is, as a number of the studies point out, crucial, with results being rendered invalid or less firm once the spatial unit is changed. Third, many of the studies take a 'snapshot' of the existing state of racial or income incidence of pollution risks, and do not ask how a general pollution-reducing policy might affect the different social groups. Fourth, as noted eariler, those that do look at both the snapshot and the directional change in inequality in light of policy measures produce some varying results. It is not always the case that, if inequality exists, it is reduced by reducing pollution generally. Fifth, and contrary to the generalisations in the literature, the studies that link risky installations with population characteristics are not at all unanimous in finding evidence of environmental inequity. Sixth, and perhaps the most important conclusion, the major part of the empirical literature makes no reference to any trade-offs in siting decisions by households. Risks may therefore differ by social group, but there is then no information on the possibility that those risks are offset by other location characteristics. The two most sophisticated studies - Millimet and Slottje (2000) and Hite (2000) suggest that such trade-offs do occur. Finally, the literature is geographically extremely biased, with all but a handful of studies coming from the USA. There is no way of knowing if the results from these studies would be replicated in other OECD countries.

7 THE BENEFITS OF ENVIRONMENTAL IMPROVEMENT: THE INCOME ELASTICITY OF 'DEMAND' FOR ENVIRONMENTAL QUALITY

Sections 5 and 6 explored the most widely studied aspect of environmental equity, namely, the extent to which poor people live in areas where environmental quality is lowest and risks highest. A recurrent theme was the differing approaches to defining inequity. The 'rights based' approach focuses on physical indicators of risk, or potential risk. The economic, or 'market dynamics' approach focuses on the extent to which individuals find themselves in situations in which the costs of environmental quality to them are out of line with the benefits to them. In this section we explore a much smaller literature which addresses the issue of whether or not environmental policy benefits the rich or the poor, respectively summarised by asking whether policy is pro-rich or pro-poor. In contrast to much of the discussion in Section 5 and 6, the issue is not whether the *status quo* condition of the environment is regressively or progressively distributed, but whether incremental change to that status quo benefits one group more than the other. There are two strands to this literature: (a) efforts to analyse the benefits (and costs) of specific pieces of legislation to assess their pro-poor or pro-rich characteristics and (b) efforts to measure the income elasticity of 'demand' for environmental quality.

7.1 Policy costs and benefits

Several studies have estimated the monetary value of environmental policy benefits, and have then sought to allocate these benefits across income groups. The relevant magnitude is then WTP_i/Y_i, where i is the ith income group. A number of studies have sought to estimate the (marginal) costs to different income groups, e.g. by looking at the tax system to see what the likely incidence is of the finance needed for the policy measure. If both benefit and cost incidence was estimated, then it would be possible to compute the net benefits of environmental policy as a fraction of income, i.e. $(WTP_i - C_i)/Y_i$. If WTP/Y is rising with income, then this is evidence that the income elasticity of WTP is greater than unity, and the good is 'pro rich' (see Section 7.2). Nonetheless, many of the original studies do not make an explicit statement about income elasticity.

7.2 Income elasticity of 'demand' for environmental quality

It is widely hypothesised that environmental quality is a 'luxury good' or an 'elitist good' so that extra provision of environmental quality will benefit the rich more than the poor (McFadden, 1994). Interestingly, this assumption has not been widely tested and appears to reflect some casual observations about the nature of the people who participate in environmental protests and organisations. To understand better how this proposition might be tested, it is necessary to investigate the notion of the income elasticity of 'demand' for environmental quality. To begin with, assume that it makes sense to speak of some 'quantity' of the environment, E. The possible relationships between E and income, Y, are shown in Figure 7, in terms of an elasticity known as the income elasticity of demand. This is defined as:

$\eta = \partial E.Y/\partial Y.E$

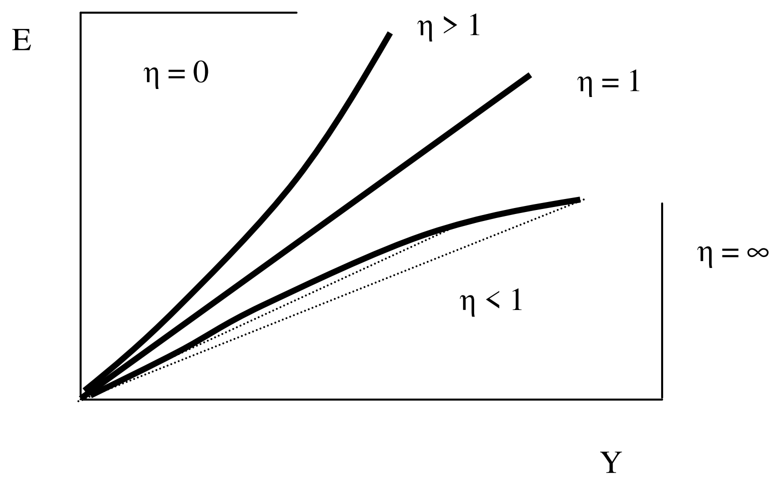

Figure 7 Income elasticities of demand

The focus of interest is on the heavy-lined curves. The extremes, where $\eta = 0$ and ∞ are shown just for comparison. It is easy to see that the income elasticities are related to another measure, namely the ratio of E to Y. For example, if $\eta < 1$, then the ratio E/Y falls as Y increases, as shown in Figure 7 by the slopes of the dotted lines. The full set of relationships is shown in Box 2.

Box 2 Income elasticities of demand and equity concepts				
Nature of good E		Income elasticity	E/Y (as Y rises)	Equity
Normal	Luxury	> 1	RISES	PRO-RICH

Normal	Unit η	= 1	CONSTANT	
Normal	Necessity	$0< \eta < 1$	FALLS	PRO-POOR
Inferior		< 0	FALLS	PRO POOR

Box 2 shows that the income elasticity of demand could be used to classify environmental goods according to whether they are pro-rich or pro-poor[26]. The basic rule is that benefits are pro-rich if the income elasticity is greater than unity (Carson et al. 2001).

Because of the nature of environmental commodities, the *quantity* demanded is often not observed. Hence it is not possible to estimate income elasticities of demand on a systematic basis. What is observed is a different elasticity, the income elasticity of willingness to pay, abbreviated to the 'income elasticity of WTP'. This magnitude is given by:

$\omega = \partial WTP.Y/\partial Y.WTP$

The relationship between the income elasticity of demand and the income elasticity of WTP is not determinate. Observations of ω do not enable us to infer observations of η. Any environmental good, E, may, for example, have a $\eta > 1$, but a ω greater than or less than unity (Flores and Carson, 1997). To see this, rearrange the equation for η as:

$Y/\partial Y = \eta. E/\partial E$

and substitute this into the equation for ω to give:

$\omega = \eta.\partial WTP.E/WTP.\partial E$

Note that $\omega < 1$ is quite compatible with $\eta > 1$, so that a good that is a 'luxury good' can have an income elasticity of WTP < 1 (Flores and Carson, 1997)[27].

Which is the relevant concept for classifying environmental goods? Both concepts convey useful information, but it has been argued that, since the focus of most environmental policy is on public goods that have some quantity constraint, the second concept, ω, is more relevant Flores and Carson, 1997)[28]. Section 8 looks at some of the evidence.

Box 3 summarises the various indicators of equity that emerge from the literature which utilises either some measure of demand, or some measure of willingness to pay.

[26] We could equally well say environmental policy that improves the environment is 'regressive' if the benefits are pro-rich and 'progressive' if benefits are pro-poor. But different writers use regressive and progressive in different ways, e.g. pro-poor benefits are sometimes called 'regressive', so pro-rich and pro-poor seem better in this context.
[27] There is a special case where there is only one public good and individuals differ in their incomes but not in preferences. Then it can be shown that $\omega = \eta/p$ where p is the price elasticity of demand (Ebert, 2000). To classify goods as pro rich or pro poor in terms of η, then, would require knowledge of ω and p.
[28] The arguments are complex, but the basic difference is that the supply of a public good is exogenous to consumers, whereas consumers choose the amount they consume of a private good. Full details can be found in Hanemann (1991) and Flores and Carson (1997).

Box 3 Summary of indicators of equity impacts of policy	
Measure	**Explanation**
E_i/Y_I	Physical 'burden' of pollution by income group
WTP_i/Y_i	Monetary value of environmental policy benefits as a fraction of income. If this rises with Y, then ω> 1
$(WTPi-C_I)/Y_i$	Monetary value of NET environmental policy benefits as a fraction of income. If this rises, then the elasticity of net WTP (consumer surplus) exceeds unity.
η	Income elasticity of DEMAND
ω	Income elasticity of WILLINGNESS TO PAY

8 THE EMPIRICAL EVIDENCE ON ENVIRONMENTAL EQUITY USING MONETISED MEASURES

8.1 Elasticity of WTP

There are two potential sources for empirical estimates of ω, the income elasticity of WTP. The first is studies of the benefits, and the costs as well, of specific policy measures. Such studies tend to estimate benefits on the basis of the *benefits transfer* technique, i.e. borrowing unit values for pollutants from other studies. The second relies on primary non-market valuation studies. The two most useful valuation techniques that permit estimation of an income elasticity are (a) travel cost and (b) contingent valuation. In each case it is necessary to identify a *valuation function*, i.e. a function linking WTP to independent variables which must include income if ω is to be estimated. Hedonic property price studies appear to be a third, but, as noted in Pearce (1980), hedonic property prices are very likely to have built into them an *assumed* income elasticity of unity. Hence, income elasticities cannot be derived from these studies. Benefits transfer studies are common but invariably are presented in a manner that makes inferring income elasticities impossible since unit values are averages taken from one or more 'primary' studies. A few studies attempt to allocate the benefits across income groups. No studies appear to be available which estimate the magnitude (WTP-C)/Y.

Evidence of the income elasticity of WTP and the income elasticity of demand is not substantial. The early literature is summarised in Pearce (1980). Kriström and Riera (1996) provides an exploratory investigation into valuation studies based on the contingent valuation approach

(CVM) to environmental goods and, more recently, Hökby and Söderqvist (2001) have assembled evidence from Swedish CVM studies. Table 4 reports estimates of ω.

Table 4 Non-market valuation benefit studies: elasticity of WTP (ω)

Study	Income elasticity of WTP	Comment
Gianessi et al. 1977: compliance with 1970 Clean Air Act, USA	0.35-0.87*	Monetised benefits by income group based on benefits transfer.
Harrison and Rubinfeld 1978: US clean air standards	1.00	Derived from an hedonic property price model and hence probably constrained to unity (see text)
Nelson 1978: noise in the US	1.00	Derived from an hedonic property price model and hence probably constrained to unity (see text)
Harris 1979: UK noise	0.20-0.40	Based on CVM
Walters 1975: UK airport noise - Heathrow - Gatwick	1.89-3.20 2.09-2.62	Suspect results due to assumption that house prices are a proxy for permanent income. Akin to an hedonic property model, using estate agents' valuations of depreciation.
Kriström and Riera, 1996 6 CVMs: Finland, France, Norway, Netherlands, Spain, Sweden	Probably less than 1	Evidence not conclusive, based on inspection of WTP equations in 6 CVM studies.
Hökby and Söderqvist (2001). 21 estimated CVM equations in Swedish valuation studies	Range -0.71 to 2.83 Mean = 0.68 Median = 0.46	CVM. One elasticity is negative. Four out of 21 elasticities > 1 16 elasticities in range 0.20-0.91
Imber et al. 1991. Kakadu conservation zone, Australia	0 0.20	Zero is the figure reported in Imber et al. but the study carries a critique by Hanemann suggesting this is incorrect. Kristrom and Riera rework the valuation equations to get 0.2
Carson et al. 1995. Exxon Valdez tanker spill in Alaska	0.28	CVM

Santos, 1998 Landscape change, UK Landscape change, Portugal Meta analysis of landscape studies	 0.20 0.30 0.57	 CVM CVM Meta analysis
Loehman and De, 1982 Avoidance of respiratory symptoms, Florida	0.26-0.60	CVM
Jones-Lee et al. 1985 Accidents	0.40	CVM
Biddle and Zarkins, 1988 Occupational risk	0.70	CVM
Viscusi and Evans, 1990 Health status	1.10	CVM
Sieg et al, 2000 Air quality, S.California	4.2-4.7	General equilibrium model of property price changes
Viscusi and Aldy, 2002 Life risks	0.5-0.6	Meta-analysis of estimates of values of statistical life
Costa and Kahn, 2002 Life risks	1.5-1.7	Time-series analysis of US value of statistical life

Notes: * estimated by author from data in the original.

One other possibility for gaining some insight into income elasticities is to look at national environmental expenditures and relate them to GNP. Expenditure is, of course, not the same as WTP, the latter exceeding the former by any consumer surplus. McFadden and Leonard (1993) suggest that, as the share of environmental expenditure tends to rise with GNP, the income elasticity must be greater than unity. Pearce and Palmer (2001) actually estimate the expenditure elasticity for European Union countries and arrive at an elasticity of 1.2, consistent with the McFadden and Leonard hypothesis[29]. Hanemann (1986) notes that national elasticities greater than unity could easily be consistent with household elasticities less than unity, especially as there is some evidence to suggest that elasticities are not constant over ranges of income.

The general impression from Table 4 is that the income elasticity of WTP for environmental change is less than unity, and numbers like 0.3-0.7 seem about right. The exception to this basic rule is the paper by Sieg et al. (2000) which, however, adopts an entirely different approach to the other studies. It involves a general equilibrium model of house price response to discrete change in air quality in Southern California. The resulting elasticities of around 4 are therefore interesting in suggesting that it is important to model distributional impacts in a general equilibrium framework. The recent work on life risks produces estimates consistent with environmental willingness to pay when meta-analysis is conducted across studies of the value of statistical life (Viscusi and Aldy, 2002), but values significantly higher than unity when time series valuations are considered (Costa and Kahn, 2002).

8.2 Elasticity of demand

[29] However, McFadden and Leonard wrongly infer from this that *any* income elasticity of *demand* must be greater than unity.

Evidence of the income elasticity of *demand* is difficult to derive from non-market valuation studies. This is because studies tend not consider contexts in which price and quantity combinations are varied. Hökby and Söderqvist (2002) pool data from several studies of the WTP for reduced marine eutrophication in the Baltic Sea. The 95% confidence interval for the estimates of η is 0.71-1.49, with a point estimate of 1.1. Using a restricted form of relationship between η and ω (see footnote 29) the value of η is 0.51.

8.3 Conclusions on income elasticity

Overall, while the evidence is limited, the general thrust of the literature is that, for individual goods, the income elasticity of *WTP* is less than unity. The recent empirical work tends to support Pearce's (1980) suggestion that the impression that environmental quality is an 'elitist' good is not justified. The implication for policy is that environmental policy is probably biased towards benefiting the poor rather than the rich.

9 THE POLICY IMPLICATIONS

Quite what policy implications follow from an analysis of income-pollution relationships will depend on how the empirical evidence is perceived. As we have seen, the evidence is :

(a) generally confined to the USA, but with some ambiguous evidence from the UK
(b) probably leaning towards the view that *existing* environmental quality and income are negatively correlated in many cases
(c) but with many caveats about the geographical generalisability of such findings and
(d) with many caveats about the extent to which any findings can be generalised across all pollutants and all environmental assets
(e) no clear findings with respect to the distributional impacts of *changes* in environmental quality, and
(f) the fairly firm finding that the income elasticity of willingness to pay is less than unity.

The main *methodological* finding has to be that by far the major part of the empirical literature on distributional incidence fails to account for any compensatory mechanisms that may exist in locational decisions. The economic theory literature is careful to point out the potential importance of such factors. For example, locating in a more polluted area produces wellbeing losses that may be, at least partially, compensated by lower prices for other goods such as housing. The *policy* relevance of this finding depends on how the distributional issue is perceived. As noted earlier, much of the environmental justice movement would not regard compensatory factors as being relevant to policy, since some form of 'equal' risk exposure is regarded as a non-tradable right. The more economically oriented approach would argue that these compensatory mechanisms need to be accounted for.

However, in so far as policy makers *are* (or should be) concerned with environmental equity, certain policy implications can be stated.

9.1 Factoring the distributional impact into decision-making

The most obvious implication is that the social incidence of policy measures needs to be factored into decision-making. On the basis that all decisions involve *some* form of comparison between costs and benefits, a decision rule would require that benefits exceed costs in the aggregate, *and* that the distributional incidence of (net) benefits should be 'acceptable'[30]. What constitutes the degree of acceptability will depend on the form of 'social welfare function' adopted by decision-makers. For example, a Rawls-type social welfare function (SWF) would sanction a policy only if its benefits accrue to the least well-off in society. Rawls (1971) argued that the distribution of resources between people is just if and only if it offers the same opportunities to all members of society. If there is inequality, resources must be distributed so that the most disadvantaged in society are favoured. This amounts to maximising the wellbeing of those who are most disadvantaged, i.e. maximising the minimum wellbeing, or 'maximin'. The 'reasonableness' of this rule is revealed by imagining that everyone is in an original state and no-one knows to which state of wellbeing they will be assigned (the 'veil of ignorance') by some policy change. Since each individual could be assigned the worst state, everyone will vote for a rule that protects the worst off. Rawls's approach is 'consequentialist' because it focuses on the outcome of a set of rules rather than on the idea of justice as 'process'[31]. A Rawlsian SWF is usually written:

$$SW = \min (U_1, U_2, \ldots U_N)$$

which means that the wellbeing of society is determined solely by the wellbeing of the individual with the lowest level of wellbeing. This SWF is strongly egalitarian. It is easy to imagine any number of variants of such rules, for example, that the greater proportion of benefits should accrue to the poor rather than the rich.

9.2 Distributional incidence and cost-benefit analysis

It is possible to 'adjust' cost-benefit analysis to allow for distributional incidence in a different way. To understand the implications it is first necessary to consider the SWF that is usually embodied in cost-benefit analysis. The 'classical utilitarian', 'purely utilitarian' or Benthamite SWF is adjusted here to allow for environment:

$$SW = U_1(x_1, e_1) + U_2(x_2, e_2) + U_3(x_3, e_3) + \ldots + U_N(x_N, e_N)$$

U = utility = welfare = wellbeing
1…N = people in society
x = quantity of consumption goods
e = quantity of environmental goods

We would expect $\frac{\partial U_i}{\partial x_i} \succ 0, \frac{\partial U_i}{\partial e_i} \succ 0$

[30] There is nothing new in this proposal. For example, it formed the basis of an extensive debate about the foundations of welfare economics in the 1950s. The *locus classicus* is Little (1950).
[31] A significant part of the EJ literature is concerned with process justice. Process approaches argue that justice is defined by agreement over the rules, regardless of what the outcome of those rules is.

This SWF assumes that there is *equal marginal utility of consumption (income)*, i.e. additions to x and e are valued equally by the various individuals. This means the social decision-maker is indifferent to *who* gets the gains (rich or poor, for example, or future generations vs present generations). This is the basic SWF underlying the general practice of cost-benefit analysis subject to a modification shortly to be introduced about what happens when there are losers as well as gainers**.** It assumes U increases with x; that U increases with e (or decreases with p); that we can measure U; that we can add up the various U_is, and that social welfare increases with individual welfare.

In practice, virtually all policies involve some people gaining and some losing (e.g. taxpayers) so that the SWF looks more like:

$$SW = U_1(x_1,e_1) + U_2(x_2,e_2) - U_3(x_3,e_3) - U_4(x_N,e_N)$$

Cost-benefit analysis embodies the Kaldor-Hicks compensation principle[32] which says that a policy is desirable it brings about a positive change in social welfare, ΔSW ($\Delta SW>0$) and that this condition is met if

$$[\Delta U_1(.) + \Delta U_2(.)] > [- \Delta U_3(.) - \Delta U_4(.)]$$

Distributional concerns can be allowed for by assigning weights to the various gains and losses to produce a 'generalisable utilitarian' SWF or 'weighted sum of utilities' SWF:

$$SW = a_1.U_1(.) + a_2.U_2(.) + a_3.U_3(.) + \ldots..+a_N.U_N(.)$$

Here the weights are given by the a's. There are various ways of deriving such weights. The first rule is simply to set

$$a_i = \frac{\bar{Y}}{Y_i}$$

where Y is income and $\bar{Y}$ is average income. The effect is to 'equalise votes' as if each person had the same average income. Thus, a poor person who has 60% of the average income would have a weight of 1/0.6 = 1.67, and a rich person with twice the average income would have a weight of 0.5. This is a crude rule that turns out to be a special case of a more general rule - see below.

The second rule sets the value of 'a' by adjusting for the elasticity of the marginal utility of income. The relevant formula is then

$$a_i = \left[\frac{\bar{Y}}{Y_i}\right]^{-\varepsilon}$$

[32] The transition obscures a switch from 'ordinal' utility in the Kaldor-Hicks world to cardinal utility. This is not discussed here.

where ε is the elasticity of the marginal utility of income function. This function links extra utility (wellbeing) to extra income and is usually assumed to take on a constant elasticity form. Since ε measures the social weight to be attached to changes in different levels of income, it is also a measure of *inequality aversion*.

The weight is shown here for individual i relative to the average income but it can be computed for any benchmark income. For example, one might set a = 1 for the richest group or person, and then express the weights on the other individuals' utility relative to this rich group (the rich person's Y would be substituted for average income in the above equation). Note that the conventional SWF in cost-benefit analysis is now a special case of this new SWF in which the a's equal unity.

What this shows is that cost-benefit analysis does not *have* to assume that the prevailing distribution of income is 'optimal'. CBA can be flexible in allowing for different SWFs.

Weighted approaches can secure very different results to 'conventional' CBA. Consider the following very simple examples.

	Gain	Loss	Net gain
Group A	+10	-4	+6
Group B	+ 2	-6	-4
Aggregate gain	+12	-10	+2

In the first example above we illustrate conventional CBA. Group A secures net gains measured by WTP of +6 but group B has net losses (perhaps measured by WTA) of -2. Overall, the gainers can compensate the losers with a net final gain of +2. CBA would approve of this policy. Even though the losers might be poor and the gainers rich.

Now let a_B = 1.6 and a_A= 1.0, then the weighted gains will be:

	Gain	Loss	Net gain
Group A	+10	-4	+6
Group B	+2x1.6 = +3.2	-6x1.6 = -9.6	-6.4
Aggregate gain	+13.2	-13.6	-0.4

The weighted approach now rejects this policy.

The value of ε is debated in the literature. An excellent survey is given by Cowell and Gardiner (1999). There it is concluded that a 'default' value of ε is unity with the range being from 0.5 to 4.0. However, values such as 4 imply a quite dramatic degree of inequality aversion. To see this consider two individuals, rich and poor, with utility functions of the form:

$$U_i = \frac{Y_i^{1-\varepsilon}}{1-\varepsilon} \qquad i = R,P$$

The ratio of the two *marginal* utilities is given by:

$$\left[\frac{Y_P}{Y_R}\right]^{\varepsilon}$$

Suppose $Y_R = 10Y_P$. The range of social values is shown below, corresponding to various values of ε[33].

ε =	0.5	0.8	1.0	1.2	1.5	2.0	4.0
Loss to R as a fraction of gain to P	0.31	0.16	0.10	0.06	0.03	0.01	~0

What this tells us is that at ε = 4, the social value of extra income to R is zero. At ε = 1, a marginal unit of income to the poor is valued ten times the marginal gain to the rich. At ε = 2, the relative valuation is 100 times. On this 'thought experiment' basis, then, values even of ε = 2 do not seem reasonable. A value of ε = 1 does seem feasible. Overall, looking at the implied values of ε in savings behaviour and at the thought-experiment above, values of ε in the range 0.5 to 1.2 seem reasonable.

9.3 Equity weighting in practice: an example

The importance of equity weighting in cost-benefit analysis can be illustrated by considering estimates of the social cost of greenhouse gases[34]. If a cost-benefit analysis of climate change control was being considered, it would be necessary to estimate the global damage done by climate change and compare it to the costs of control. A survey of the estimates of damage can be found in Pearce (2002). Since a disproportionate share of the damages accrues to developing countries (relative to GNP) there is a strong case for equity weighting. To illustrate how equity weighting affects global damage estimates, we employ the SWF introduced previously:

$$D_{WORLD} = D_R \cdot \left[\frac{\bar{Y}}{Y_R}\right]^{\varepsilon} + D_P \cdot \left[\frac{\bar{Y}}{Y_P}\right]^{\varepsilon}$$

where R = rich and P = poor and D is the monetary value of damage. Crude estimates of the relevant magnitudes are then D_R = \$216 billion and D_P = \$106 billion, for 2 x CO_2 (Fankhauser, 1995); Y_R = \$10,000, and Y_P= \$1110; and $\bar{Y}$ = \$3333[35]. Substituting in [9] produces estimates of world damage of

[33] The ratio of incomes between R and P has been chosen to illustrate *international* differences in real income per capita. The ratio would be far smaller for analysis of a policy *within* an OECD country. Across OECD countries the ratio could reach 7, the ratio for the USA compared to Mexico.

[34] Note that the value of ε enters a cost-benefit analysis in two ways: as a measure of inequality aversion across different income groups if the costs and benefits are equity weighted, and as a component of the social time preference rate as a discount rate, i.e. inequality aversion through time. This underlines the importance of choosing the 'correct' estimate of ε.

[35] We take rich countries to be OECD countries, poor to be everyone else.

unweighted	\$ 322 billion
weighted, $\varepsilon = 0.5$	\$ 307 billion
weighted, $\varepsilon = 0.8$	\$ 343 billion
weighted, $\varepsilon = 1$	\$ 390 billion
weighted, $\varepsilon = 1.5$	\$ 600 billion

It can be seen that the value of ε matters a great deal. If $\varepsilon = 0.5$, there is little change to the unweighted estimates of damage. If $\varepsilon=1$, there is a 20% increase in global damages, and if $\varepsilon=1.5$ damages rise by nearly 100%. If damages are doubled, then the benefits from avoiding climate change are also doubled, with formidable implications for the amount of action that would be taken to control climate change.

9.4 Cost-benefit analysis and income elasticity of WTP

Sections 7 and 8 discussed the notion of the income elasticity of willingness to pay, ω. Leaving aside the issue of equity weighting discussed in Section 9.2, the value of ω can be extremely important in practical cost-benefit analysis. For example, to appraise an investment in environmental conservation requires not only that the benefits be estimated for the near future but over the longer run as well. But over the longer run population may grow, in which case the total benefits of the conservation investment will grow[36] at the rate of population growth. This effect is often not factored into actual cost-benefit studies. But if incomes grow with time then WTP is also likely to grow: indeed, this is what an income elasticity of WTP measures. Hence there needs to be a second adjustment to the benefit stream. The final formula to account for population effects and income elasticity is:

$$\frac{B_t}{B_{t-1}} = [1 + \omega.\dot{y} + \dot{p}]$$

where the expression on the left hand side is the growth rate of total benefits, $\dot{y}$ is the rate of growth of per capita income, and $\dot{p}$ is population growth. To illustrate the effects, assume population growth is zero, that income grows at 3% per annum and that ω is 0.3, a result that is consistent with the empirical survey in Section 8. Then benefit growth would be 0.3x0.03 = 0.01 or one per cent per annum. For an investment with a 50 year time horizon, the effect would be to add 65% to the estimated benefits.

9.5 Fiscal policy and environmental distribution

Environmental damage may be successfully tackled through policy measures such as environmental taxes and regulations. However, those measures may themselves have distributional implications. Thus poorer households may not only suffer more exposure to environmental harm, but the means of reducing the harm may also impact more heavily on them

[36] Assuming the asset is a public good and that there are no 'congestion effects', i.e. the greater the number 'consuming' the good, the lower is the wellbeing of the existing users as the number of users expands. Strictly, such good s are 'club good' as opposed to public goods.

relative to their incomes. Clearly, there are various potential combinations of the distribution of harm and the distribution of the policy costs. In the worst case, damage may be regressively distributed and policy costs may also be regressively distributed. If damage is 'progressive', i.e. suffered mainly by the rich, a similar equity issue will arise if the poor pay disproportionately more than the rich to resolve the problem. In such circumstances, policies need to address both the cost and benefit side of the picture. As indicated earlier, many OECD governments already embody distributional concerns in their development of fiscal measures for the control of environmental damage, with various measures being used to lower the cost incidence on the poor: rebates, zero charges, reduced charges, compensation measures etc. The central conclusion is that addressing the distributional incidence of environmental damages (benefits) is not sufficient. Care has to be taken also to address the incidence of the policy costs as well.

9.6 **Summary of policy implications**

The policy implications of the analysis of distributional incidence issues can be summarised as follows:

(a) the social distribution of *existing risks* gives rise to equity concerns that many would argue need to be remediated by targeted action to improve environments in areas where low income and vulnerable groups exist. These remediation policies may or may not be influenced by the extent to which environmental risks are offset by other gains from locating in higher risk areas.

(b) the social incidence of *new policy measures* is a legitimate cause for concern in decision-making. At the very least, an analysis of who gains and loses from policy measures is required. Methodologies exist for adjusting conventional cost-benefit criteria to account for equity impacts. Depending on the accepted measure of inequality aversion within a society or across nations, the effects of equity weighting can be substantial. It is important to assess the distributional incidence of the benefits <u>and</u> the costs of environmental policy.

(c) the social incidence literature produces estimates of the income elasticity of willngness to pay for environmental improvement. Contrary to popular belief, this income elasticity is almost certainly below unity. Such income elasticities need to be factored into cost-benefit studies of environmental impacts. Again, the choice of the 'right' value can have significant effects on the outcome of a cost-benefit appraisal.

(d) Care has to be taken that active discouragement of the siting of polluting activities in low income or ethnic minority areas does not harm employment prospects in those areas. The views of people in the relevant areas need to sought. This places a premium on policies that generate full information about development prospects and their associated risks: the poor tend to have less access to information compared to the rich. One risk is that some 'non-users' will be active politically in trying to prevent developments from being sited in particular areas. Care needs to be taken to ensure that these activists are representative of local people who are directly affected and who face the real trade-off between development and environmental risk. There may also be detrimental environmental effects – e.g. 'brownfield' sites might be avoided because of a fear of environmental injustice, with the result that new development shifts to Greenfield sites where amenity values may be very high. In the same vein, policies designed to improve the environments of poor areas may force up rents and property prices, giving rise to longer term forces that encourage the poor to move out of the area and the rich to move in ('gentrification'). In short, 'pro poor' policies need to be evaluated carefully for their second and third round effects.

(e) Finally, where distributional effects are significant and a matter of serious concern, some of the proceeds from environmental taxes may be earmarked for allocation to improvement policies in poor areas.

REFERENCES

Anderson, A., Anderton, D and Oakes, J. 1994. Environmental equity: evaluating TSDF siting over the past two decades, *Waste Age*, **25**, 83-100

Anderton, D., Anderson A., Oakes, J., Fraser, M, Weber, E and Calabrese, E. 1994a. Hazardous waste facilities: environmental equity issues in metropolitan areas, *Evaluation Review*, **18**, 2, 123-140

Anderton, D., Anderson A., Oakes, J., Fraser, M. 1994b. Environmental equity. The demographics of dumping in Dixie. *Demography*, **31**, 2, 229-248

Arora, S and Cason, T. 1999. Do community characteristics influence environmental outcomes? Evidence from the Toxic Releases Inventory. *Southern Economic Journal*, **65**, 691-716

Asch, P and Seneca. J. 1978. Some evidence on the distribution of air quality. *Land Economics*, **54.** 218-257

Atkinson, G., Machado, F and Mourato, S. 2000. Balancing competing principles of environmental equity. *Environment and Planning A*. **32**. 1791-1806

Atlas, M. 2002. *Rush to Judgement: An Empirical Analysis of Environmental Equity in US Environmental Protection Agency Enforcement Actions*, Department of Political Science and Public Administration, North Carolina State University, mimeo.

Atlas, M. 2001. Safe and sorry: risk, environmental equity, and hazardous waste management facilities. *Risk Analysis*, **21**, 939-954

Bae, C-H 1997. How Los Angeles' air quality policies benefit minorities, *Journal of Environmental Planning and Management*,**40** (2), 235-260

Bateman, I., Jones, A., Lovett, A., Lake I and B.Day. 2002. Applying geographical information systems to environmental and resource economics, *Environmental and Resource Economics*, **22**, (1-2), 219-269

Bateman, I., Jones, A., Lovett, A., Lake I and B.Day. 2003. Modelling environmental equity: Exposure to environmental urban noise pollution in Birmingham, UK. *Urban Studies*, forthcoming.

Baumol, W. 1972. Environmental protection and the distribution of incomes, in OECD, *Problems of Environmental Economics*, Paris: OECD,

Baumol, W. 1974. Environmental protection and income distribution. In H.Hochman and G.Peterson (eds), *Redistribution through Public Choice*, New York: Columbia University Press, 93-104.

Baumol, W and Oates, W. 1988. *The Theory of Environmental Policy*, 2nd edition. Cambridge: Cambridge University Press.

Becker, G. 1983. A theory of competition among pressure groups for political influence. *Quarterly Journal of Economics*, **98**, 371-400

Been, V. 1993. What's fairness got to do with it? Environmental justice and siting of locally undesirable land uses. *Cornell Law Review*, **78**, 1001-1085

Been, V. 1994. Locally undesirable land uses in minority neighbourhoods. Disproportionate siting or market dynamics. *Yale Law Journal*, **10**, 1383

Been, V and Gupta, F. 1997. Coming to the nuisance or going to the barrios? A longitudinal analysis of environmental justice claims. *Ecology Law Quarterly*, **24**, 1-56

Berry, N. *The Social Burdens of Environmental Pollution*. Cambridge MA: Ballinger

Biddle, J and Zarkin, G. 1988. Worker preferences and market compensation for job risk. *Review of Economics and Statistics*, **70**, 4, 660-666

Boer, J., Pastor, M, Sadd, J and Snyder, L. 1997. Is there environmental racism? The demographics of hazardous waste in Los Angeles County. *Social Science Quarterly*, **78**, 793-810

Bowen, W., Salling, M., Haynes, F and Cyran, E. 1995. Toward environmental justice - spatial equity in Ohio and Cleveland. *Annals of the American Association of Geographers*. **85**, 14, 623-640

Bowen, W. 2002. An analytical review of environmental justice research: what do we really know? *Environmental Management*, **29** (1), 3-15

Blomquist, G., Berger, M and Hoehn, J. 1988. New estimates of quality of life in urban areas, *Amercian Economic Review*, **78**, 1, 89-107

Brainard, J., Jones, A., Bateman, I., Lovett, A and P.Fallon. 2002. Modelling environmental equity: access to air quality in Birmingham, UK. *Environment and Planning A*.

Brainard, J., Jones, A., Bateman, I., Lovett, A and Day, B. 2003a. *Modelling Environmental Equity: Exposure to Environmental Noise Pollution in Birmingham*, UK. Paper to OECD Workshop on the Distribution of Benefits and Costs of Environmental Policies: Analysis, Evidence and Policy Issues. Paris: OECD.

Brainard, J., Bateman, I and Jones, A. 2003b. Accessing urban park locations: contrasting equity of access with economic efficiency measures. University of East Anglia, CSERGE, *mimeo*.

Brajer, V and Hall, J. 1992. Recent evidence on the distribution of air pollution effects, *Contemporary Policy Issues*, **X**, April, 53-71

Bullard, R.D. 1983. Solid waste sites and the Black Houston community, *Sociological Inquiry*, **53**. 273-288

Burke, L. 1993. Race and environmental equity: a geographic analysis in Los Angeles, *Geographic Information Systems*. October, 44-50.

Cardenas, J., Stranlund, J and Willis, C. 2002. Economic inequality and burden sharing in the provision of local environmental quality, *Ecological Economics*. **40**, 379-395

Carraro, C. 2002. Climate change policy: models, controversies and strategies, in T Tietenberg and H Folmer (eds), *The International Yearbook of Environmental and Resource Economics 2002/2003*, Cheltenham: Edward Elgar, 1-65

Carson, R., Mitchell, R, Hanemann, M., Kopp, R., Presser, S and Ruud, P. 1995. *Contingent Valuation and Lost Passive Use Damages from the Exxon Valdez*. Discussion Paper 95-02. Department of Economics, University of San Diego

Carson, R., Flores, N and Meade, N. 2001. Contingent valuation: controversies and evidence. *Environmental and Resource Economics*, **19**, 173-210

Chakraborti, J. 2001. Acute exposure to extremely hazardous substances: an analysis of environmental equity, *Risk Analysis*, **21**, 5, 883-894

Coase, R. 1960. The problem of social cost. *Journal of Law and Economics*, **3**, 1-44.

Costa, D and Kahn, M. 2002. *Changes in the Value of Life 1940-1980*. Working Paper 9396, Cambridge MA: National Bureau of Economic Research. www.nber.org/papers/w9396.

Cowell, F and Gardiner, K. 1999. *Welfare Weights*. Report to the UK Office of Fair Trading. Available at www.oft.gov.uk/NR/rdonlyres/

Cutter, S. 1995. Race, class and environmental justice. *Progress in Human Geography*. **19** (1), 111-122.

Cutter, S., Holm,D and Clark, L. 1996. The role of geographic scale in monitoring environmental justice. *Risk Analysis*, **16**, 517-526

de Kam, F. 2002. Issues Paper, In OECD, *Implementing Environmental Fiscal Reform: Income Distribution and Sectoral Competitiveness Issues.* COM/ENV/EPOC/DAFFE/CFA(2002)76Final. Paris, OECD.

Ebert, U. 2000. *Environmental Goods and the Distribution of Income*, Department of Economics, Discussion Paper V-199-00. Oldenbourg: University of Oldenbourg.

Fankhauser, S. 1995. *Valuing Climate Change: the Economics of the Greenhouse*. London: Earthscan.

Flores, N and Carson, R. 1997. The relationship between income elasticities of demand and willingness to pay. *Journal of Environmental Economics and Management*, **33**, 287-295

Foreman, C. 2000. Environmental justice and risk assessment: the uneasy relationship. *Human and Ecological Risk Assessment*, **6** (6), 549-554

Freeman, A.M. 1972. The distribution of environmental quality. In A.Kneese and B.Bower (eds), *Environmental Quality Analysis*. Baltimore: Johns Hopkins UP, 243-80

Friends of the Earth. 2001. *Pollution and Poverty - Breaking the Link*. London: Friends of the Earth

GAO (US General Accounting Office) 1983. *Siting of Hazardous Waste Landfills and their Correlation with Racial and Economic Status of Surrounding Communities*. Washington DC: US GAO

Garrod, G and Willis, K. 1999. *Economic Valuation of the Environment: Methods and Case Studies*. Cheltenham: Edward Elgar

Gianessi, L., Peskin, H and Wolff, E. 1977. The distributional implications of national air pollution damage estimates. In F Juster (ed), *The Distribution of Economic Wellbeing*. Cambridge Mass: Ballinger. 201-226

Glickman, T and Hersh, R. 1995. *Evaluating Environmental Equity: The Impacts of Industrial Hazards on Selected Social Groups in Allegheny County, Pennsylvania*, Discussion Paper 95-13, Washington DC: Resources for the Future.

Goldman, B. 1994.*Not Just Prosperity: Achieving Sustainability with Environmental Justice*. Washington DC: National Wildlife Federation.

Goldman, B. 2000. An environmental justice paradigm for risk assessment, *Human and Ecological Risk Assessment*, **6** (6), 541-548

Goldman, B and Fitton, L. 1994. *Toxic Wastes and Race Revisited: an Update of the 1987 Report on the Racial and Socioeconomic Characteristics of Communities with Hazardous Waste Sites*. Washington DC: Center for Policy Alternatives.

Hamilton, J. 1993. Politics and social costs: estimating the impact of collective action on hazardous waste facilities. *RAND Journal of Economics*, **24**, 1, 101-125

Hamilton, J. 1995. Testing for environmental racism: prejudice, profits, political power ? *Journal of Policy Analysis and Management*. **14**, 107-32

Hamilton, J. 1999. Exercising property rights to pollute: do cancer risks and politics affect plant emission reductions? *Journal of Risk and Uncertainty*, **18**, 2, 105-24

Hamilton, J. 2003. *Environmental Equity and the Siting of Hazardous Waste Facilities in OECD Countries: Evidence and Policies*. Paper to OECD Workshop on the Distribution of benefits and Costs of Environmental Policies: Analysis, Evidence and Policy Issues. Paris: OECD.

Hamilton, J and Viscusi, W.K. 1999. *Calculating Risks? The Spatial and Political Dimensions of Hazardous Waste Policy*. Cambridge MA: MIT Press

Handy,F. 1977. Income and air pollution in Hamilton, Ontario. *Alternatives*. **6**. 18-24

Hanemann, M. 1991. Willingness to pay and willingness to accept: How much can they differ? *American Economic Review*, **81**, 635-647

Hanemann, M. 1996. Theory versus data in the contingent valuation debate. In D.Bjornstad and J.Kahn (eds), *The Contingent Valuation of Environmental Resources: Methodological Issues and Research Needs*. Cheltenham: Edward Elgar, 38-60

Harbaugh, W., Levinson, A and Wilson, D. 2000. *Re-examining the Empirical Evidence for an Environmental Kuznets Curve*, Working Paper 7711, National Bureau of Economic Research, Cambridge, Mass.

Harris, A. 1979. Is quiet a luxury good? - a survey approach. *International Journal of Social Economics*, **6**, 177-188

Harrison, D. 1975. *Who Pays for Clean Air?* Cambridge MA: Ballinger

Harrison,K and Antweiler, W. 2002. Incentives for pollution abatement: regulations, regulatory threat and non-governmental pressure. University of British Columbia, mimeo.

Harrison, D and Rubinfeld, D. 1978. Hedonic house prices and the demand for clean air. *Journal of Environmental Economics and Management*, **5**, 81-102

Haynes, K., Lall, S and Trice, M. 2001. Spatial issues in environmental equity. *International Journal of Environmental Technology and Management*, **1**, 1, 17-31

Hite, D. 2000. A random utility model of environmental equity, *Growth and Change*, **31**, 40-58

HM Treasury (UK), 2002. *The Green Book: Appraisal and Evaluation in Central Government* (Draft). London: HM Treasury. www.hm-treasury.gov.uk

Hökby, S and Söderqvist, T 2002. *Elasticities of Demand and Willingness to Pay for Environmental Services in Sweden,* Paper to 11th Annual Conference of the European Association of Environmental and Resource Economists, Southampton, 2001.

Hockman, E and Morris, C. 1998. Progress towards environmental justice: a five- year perspective on toxicity, race and poverty in Michigan, 1990-1995. *Journal of Environmental Planning and Management*, **41**, 2, 157-176

Imber, D., Stevenson G and Wilks, L. 1991. *A Contingent Valuation of the Kakadu Conservation Zone*, Resource Assessment Commission, Research Paper No.3, Canberra

Jerrett, M., Eyles, J., Cole D and S.Reader. 1997. Environmental equity in Canada: an empirical investigation into the income distribution of pollution in Ontario. *Environment and Planning A*. **29**. 1777-1800

Jones-Lee, M., Hammerton, M and Philips, P. 1985. The value of safety: the results of a national sample survey. *Economic Journal*, **95**, 49-72.

Kahn, M. 1998. A household level environmental Kuznets curve. *Economics Letters*. **59**. 269-273

Kahn, M. 2001. The beneficiaries of Clean Air Act regulation. *Regulation*, Spring, 34-38

Khanna, N. 2001. *The Income Elasticity of Non-Point Source Air Pollutants*. Working Paper WP0110. Economics Department, New York: Binghampton University

Kriesel, W., Centner, T and Keeler, A. 1996. Neighborhood exposure to toxic releases: are there racial inequities? *Growth and Change*, **27**, 477-499

Kriström, B and Riera, P. 1966. Is the income elasticity of environmental improvements less than one? *Environmental and Resource Economics*, **7**, 45-66

Kruize, M and Bouwman, A. 2003. *The Distribution of Benefits and Costs of Environmental Policies: a Case Study on the Distribution of Environmental Impacts in Rijnmond Region: The Netherlands.* Paper to OECD Workshop on the Distribution of benefits and Costs of Environmental Policies: Analysis, Evidence and Policy Issues. Paris: OECD.

Lavelle, M and Coyle , M. 1992. Unequal protection: the racial divide on environmental law, *National Law Journal*, September 21, S1

Little, I.M.D. 1950. *A Critique of Welfare Economics*. Oxford: Oxford University Press.

Liu, F. 2001. Dynamics and causation of environmental equity, locally unwanted land uses, and neighbourhood changes. *Environmental Management*, **21**, 5, 643-656

Loehman, E and De, E. 1982. Application of stochastic choice modeling to policy analysis of public goods: a case study of air quality improvements. *Review of Economics and Statistics*, **64**, 3, 474-80

Loomis. J and Walsh, R. 1997. *Recreation Economic Decisions: Comparing Benefits and Costs*. Venture Publishing, State College, Pennsylvania

McFadden, D. 1994. Contingent valuation and social choice. *American Journal of Agricultural Economics*, **76**, 689-708

McFadden, D and Leonard, G. 1993. Issues in the contingent valuation of environmental goods: methodologies for data collection and analysis. In J.Hausman (ed), *Contingent Valuation: a Critical Assessment*. Amsterdam: Elsevier

McCleod, H., Langford, I., Jones, A., Stedman, J., Day, R., Lorenzoni, I and Bateman, I. 2000. The relationship between socio-economic indicators and air pollution in England and Wales. *Regional Environmental Change*, **1** (2), 78-85

Millimet, D and Slottje, D. 2000. *The Distribution of Pollution in the US: an Environmental Gini Approach*. Department of Economics, Southern Methodist University, Dallas, *mimeo*.

Mohai, P and Bunyan, B. 1992. Environmental racism. In P.Mohai and B.Bunyan (eds), *Race and the Incidence of Environmental Hazards: A Time for Discourse*. Boulder: Westview Press, 163-176

Nelson, J. 1978. Residential choice, hedonic prices and the demand for urban air quality. *Journal of Urban Economics*, **5**, 357-369

OECD. 2002. *Distributional Effects of Environmental Policy: Roundtable Discussion on Assessments and Measures Taken in Member Countries – Issues Paper*. ENV/EPOC/WPNEP(2002)20. Paris: OECD

Olson, M. 1965. *The Logic of Collective Action*. Cambridge MA: Harvard University Press.

Pearce, D.W. 1980. The social incidence of environmental costs and benefits. *Progress in Resource Management and Environmental Planning*, **2**, 63-87

Pearce, D.W. 2002. The social cost of carbon and its policy implications. In D.Helm and D.W.Pearce (eds), *Controlling Global Warming*, forthcomimg.

Pearce, DW and Turner, R. 1990. *Economics of Natural Resources and the Environment*, Hemel Hempstead: Wheatsheaf.

Pearce, DW and Palmer, C. 2001. Public and private spending for environmental protection: a cross country policy analysis, *Fiscal Studies*, **22**, 4, 403-456

Pellow, D., Weinberg, A and Schnaiberg, A. 2001. The environmental justice movement: equitable allocation of the costs and benefits of environmental management outcomes. *Social Justice Research*, **14**, 4, 423-439

Perlin, S., Setzer, R W., Creason, J and Sexton, K. 1995. Distribution of industrial air emissions by income and race in the United States: an approach using the Toxic Release Inventory, *Environmental Science and Technology*, **29**, 69-80

Pye, S., Stedman, J., Adams, M and King, K. 2001. *Further Analysis of NO_2 and PM_{10} Air Pollution and Social Deprivation*. Report AEAT/ENV/R/0865. Culham: AEA Technology

Rawls, J. 1971. *A Theory of Justice*. Oxford: Oxford University Press

Ringquist, E. 1998. A question of justice: equity in environmental litigation, *The Journal of Politics*, **60**, 4, November, 1148-65

Roberts, S. 2000. Environmental justice: examining the role of risk assessment. *Human and Ecological Risk Assessment*, **6** (6), 537-540

Rose, A.,Stevens, B and David G. 1989. Assessing who gains and who loses from natural resource policy: distributional information and the public participation process. *Resources Policy*, **15**, 4, 282-291

Santos, J.M.L. 1998. *The Economic Valuation of Landscape Change: Theory and Policies for Land Use and Conservation*. Cheltenham: Edward Elgar

Sexton, K. 2000. Socioeconomic and racial disparities in environmental health: is risk assessment part of the problem or part of the solution? *Human and Ecological Risk Assessment*, **6** (6), 561-574

Sieg, H., Smith, V.Kerry., Banzhaf, H.S and Walsh, R. 2000. Estimating the general equilibrium benefits of large policy changes: the Clean Air Act revisited. National Bureau of Economic Research, Working Paper 7744. Cambridge MA: NBER.

Simon, T. 2000. In defense of risk assessment: a reply to the environmental justice movement's critique. *Human and Ecological Risk Assessment*, **6** (6), 555-560

Smith, K. 1988.Air pollution: assessing total exposure in developing countries, *Environment*, **30**, 10, 16-20, 28-35

Tiebout, C. 1956. A pure theory of local expenditures, *Journal of Political Economy*, **LXIV**, 416-24.

UCC (United Church of Christ), 1987. *Toxic Wastes And Race In The United States. A National Report On The Racial And Socio-Economic Characteristics With Hazardous Waste Sites*. New York: UCC - Commission on Racial Justice

US EPA (Environmental Protection Agency). 1992. *Environmental Equity: Reducing Risk for All Communities*. EPA/230-R-92-008. Washington DC: US EPA

US EPA (Environmental Protection Agency). 1998. *Final Guidance for Incorporating Environmental Justice Concerns in EPA's NEPA Compliance Analysis*. Washington DC: Government Printing Office

US GAO (General Accounting Office).1983. *Siting of Hazardous Waste Landfills and their Correlation with Racial and Economic Status of Surrounding Communities*. Washington DC: GAO

Viscusi, W.K and Evans, W. 1990. Utility functions that depend on health status: estimates and economic implications. *American Economic Review*, **80**, 3. 353-74

Viscusi, W.K and Aldy, J. 2002. *The Value of Statistical Life: a Critical Review of Market Estimates throughout the World.* Discussion Paper 392. John M Olin Center fir Law, Economics and Business. Cambridge MA: Harvard University

Wagstaff, A and van Doorslaer, E. 2000. Income inequality and health: What does the literature tell us? *Annual Review of Public Health*, **21**, 543-567

Walker, G., Fairburn, J and Bickerstaff, K. 2000. *Ethnicity and Risk: the Characteristics of Populations in Census Wards Containing Major Accident Hazard Sites in England and Wales*. Occasional Paper 15. Department of Geography, University of Staffordshire

Walters, A. 1976. *Noise and Prices*. Oxford: Oxford University Press

Wildavsky, A. 1995. *But Is It True? A Citizen's Guide to Environmental Health and Safety Issues*. Cambridge, Mass: Harvard University Press.

Yandle, T and Burton, D. 1996. Re-examining environmental justice: a statistical analysis of historical hazardous waste landfill siting patterns in metropolitan Texas. *Social Science Quarterly*, **77**, 477-492

Young, H Peyton. 1994. *Equity in Theory and Practice*, Princeton: Princeton University Press

Zajac, E. 1995. *The Political Economy of Fairness*. Cambridge, Mass: MIT Press.

Zimmerman, R. 1993. Social equity and environmental risk. *Risk Analysis*, **13**, 649-666

Zupan, J. 1973. *The Distribution of Air Quality in the New York Region*. Baltimore: Johns Hopkins UP.