View metadata, citation and similar papers at core.ac.uk
brought to you by CORE
provided by Research Papers in Economics

The Distributional Effects of Environmental Tax Reform

Nick Johnstone and Janaki Alavalapati

Environmental Economics Programme

Discussion Paper

DP 98-01

October 1998

International Institute for Environment and Development

IIED is an independent, non-profit organisation which seeks to promote sustainable patterns of world development through research, policy studies, consensus building and public information. Established in 1971, the Institute advises policy makers and supports and collaborates with southern specialists and institutions working in similar areas.

Environmental Economics Programme

The Environmental Economics Programme (EEP) seeks to develop and promote the application of economics to environmental issues in developing countries. This is achieved through research and policy analysis on the role of the environment and natural resources in economic development and poverty alleviation.

The Authors

Nick Johnstone is a Research Associate in the Environmental Economics Programme, International Institute for Environment and Development, London.

Janaki Alavalapati is Assistant Professor at the School of Forest Resources and Conservation, University of Florida, Gainesville.

Correspondence should be addressed to Nick Johnstone at:

Environmental Economics Programme
IIED
3 Endsleigh Street
London WC1H 0DD
UK

Tel: **(44) 0171 388 2117**
Fax: **(44) 0171 388 2826**
Email: **nick.johnstone@iied.org**

Acknowledgements

This work was supported through the generous financial assistance of WWF-UK.

THE DISTRIBUTIONAL EFFECTS OF ENVIRONMENTAL TAX REFORM

Nick Johnstone and Janaki Alavalapati

ABSTRACT

In recent years there has been increased debate about the potential for shifting the incidence of the tax system away from a variety of economic 'goods' (i.e. employment, investment, etc ...) and towards environmental 'bads' (i.e. pollution emissions, resource extraction, etc ...). However, in spite of their apparent efficiency, economic instruments have been adopted relatively less frequently than direct regulation to mitigate environmental damages. One reason may be that some of the distributional implications of environmental tax reform have not been adequately recognised and addressed. How the costs and benefits of environmental policies are distributed in society is critical for their application since this will play a significant role in determining whether or not a particular measure is likely to be politically feasible. Moreover, for a given level of aggregate economic wealth, a redistribution of resources from richer households toward poorer households will tend to increase overall social welfare, and *vice versa*. While environmental measures should not be the instrument through which distributional objectives are realised, their growing importance means that distributional implications can no longer be ignored, particularly in the face of increasing economic inequality in many countries.

This report reviews some of the distributional implications of environmental tax reform in the residential energy, road transport and agriculture sectors. While some of the most important distributional issues are related to the direct financial burden of the tax, this study also reviews some of the other distributional implications. In particular, it looks at the indirect effects on goods and services through input-output linkages, the potentially mitigating effects through different forms of revenue recycling, the distribution of indirect economic effects such as employment opportunities, as well as the distribution of social and environmental effects such as personal health and exposure to pollutants.

The paper argues that in many cases the distributional consequences of environmental tax reform may be distinctly regressive, at least in terms of relative tax burdens. The distribution of environmental and social consequences are much less readily quantifiable, but in many cases their effects may be progressive. However, this depends very much on the sector affected and the precise form of the reform introduced. In addition, the revenue raised by environmental taxes (unlike most other environmental policy measures) provide the means whereby some of these adverse distributional consequences can be mitigated and even reversed. Finally, by addressing market failures and barriers which impact particularly upon lower-income households some measures which mitigate the adverse distributional effects of

environmental tax reform can also improve the economic efficiency of the reform. Thus, if designed appropriately, environmental tax reform can meet both distributional and environmental objectives in an efficient manner.

On the basis of the evidence reviewed it is concluded that distributional concerns, while important in many cases, should not prevent or delay the introduction of environmental taxes. Rather, they should serve as guiding principles in the design of environmental tax reform not only for their own sake, but also because efficiency objectives and equity objectives can be complementary in a well-designed package of environmental tax reform.

TABLE OF CONTENTS

I. Introduction 1

II. Environmental Tax Reform: Examples and Extent 3

II.A The Case for Environmental Taxes 4

II.B The Nature of Environmental Tax Reform 5

II.C The Extent of Environmental Tax Reform *6*

III. The Distribution of the Financial Effects of Environmental Taxation 7

III.A Expenditures on Environmentally-Sensitive Goods by Income Group 7

III.B Behavioural Responses to Price Changes by Income Group 14

III.C Revenue Recycling and Mitigating Policies 18

III.D Conclusion 21

IV. The Distribution of the Environmental and Social Effects of Environmental Tax Reform 21

IV.A Distribution of Environmental Effects of ETR 21

IV.B Distribution of Social Effects of ETR 26

IV.C Conclusions 33

V. Reconciling the Economic, Environmental and Social Effects of Environmental Tax Reform 33

VI. Conclusion 36

VII. References 38

The Distributional Effects of Environmental Tax Reform

Nick Johnstone and Janaki Alavalapati[1]

I. Introduction

In recent years there has been increased debate about the potential for shifting the incidence of the tax system away from a variety of economic 'goods' (i.e. employment, investment, etc ...) and towards environmental 'bads' (i.e. pollution emissions, resource extraction, etc ...). (See Repetto 1992 for a discussion. O'Riordan 1997 provides a more recent review of some of the issues involved.) In its most comprehensive form, this has been described as 'ecological (or environmental) tax reform,' (ETR) in which the changes are sufficient to bring about a fundamental restructuring of the economy along less environmentally damaging lines.

To some extent this debate is a logical extension of the economic analysis of environmental externalities, since economists have long argued for the use of economic instruments (environmental taxes and permits) against command and control policies as a means of managing environmental resources (natural resource stocks and environmental quality). (See Baumol and Oates 1988.) This argument is usually made on the grounds of the perceived relative economic efficiency of the former. However, in spite of their apparent efficiency, economic instruments have been adopted relatively less frequently than direct regulation to mitigate environmental damages. (See OECD 1994a for a review.)

One reason may be that some of the distributional issues involved in a widespread shift in the incidence of taxation have been neglected.[2] How the costs and benefits of environmental policies are distributed in society is critical for their application for two related reasons:

- First, the distribution of costs and benefits will play a significant role in determining whether or not a particular measure is likely to be politically feasible; and,

- Second, for a given level of aggregate economic wealth, a redistribution of resources from richer households towards poorer households will tend to increase overall social welfare, and vice versa.[3]

While environmental measures should not be the instrument through which distributional objectives are realised, their growing importance means that distributional implications can no longer be ignored, particularly in the face of increasing economic inequality in many countries. For instance, in the UK between 1981-1983 and 1991-1993 the percentage income share of the bottom decile fell from 4.1% to 2.9%, while the figure for the top decile rose from 21.3% to 26.2% (Goodman, Johnson and Webb 1997). The Gini Coefficient,

[1] Helpful comments from Nick Mabey and Josh Bishop on initial drafts of the paper are gratefully acknowledged. Thanks also to Frances Reynolds who assisted in the preparation of the document.

[2] This omission is hardly surprising since it was usually assumed that the changes would be marginal.

[3] See Boadway and Bruce (1994) for a discussion of the theoretical conditions under which this is likely to be the case.

an index of inequality, rose from 0.25 in 1977 to 0.34 in 1993.[4] Thus, besides their relative efficiency, policy makers need to know the distributional impacts of different instruments among different groups of people in order to determine whether or not such a policy change is both feasible and desirable. Moreover, as the use of economic instruments as an environmental policy tool increases, they will in themselves come to be significant determinants of the overall distribution of resources within an economy.

Unfortunately, determining the distributional effects of ETR in terms of overall welfare is exceedingly complex. This can be illustrated with reference to a petrol tax. In the first instance, the tax will affect the *relative tax burden* for different groups in society.[5] This will arise both directly through differences in consumption of petrol, and indirectly through the 'petrol-intensity' of other goods consumed. However, a petrol tax will raise significant revenue, and thus it is important also to examine the incidence of the *benefits from recycling the revenue* generated, whether through decreases in other taxes or increased public expenditure. In addition, the incidence of more *indirect economic effects* on different groups may also be important. For instance, a tax on petrol would probably hurt those involved in the automobile and ancillary industries (as employees or shareholders). The incidence of *social effects* may also vary significantly, with some households benefiting from reduced noise levels due to lower traffic volume, while other households face restricted mobility. And finally, the distributional effects of a petrol tax will also be reflected in the incidence of *environmental damages*, such as exposure to pollution.[6]

This report reviews these different effects for three sectors: residential energy, private road transport and agriculture. These sectors have been chosen because they are generally regarded as environment-intensive, both in terms of resource use and in terms of pollution emissions. (See Table 1.) They are also important targets of environmental taxes, and are likely to be increasingly so in the future. (See OECD 1994a for a review of the application of environmental taxes by sector.) And finally, they are sectors in which expenditures by consumers are quite direct, rather than arising through inputs into other sectors. This makes it easier to trace the distributional effects of policy measures aimed at emissions from the sector. The discussion focuses on differences in the distribution of costs and benefits for different income and/or expenditure groups.[7]

[4] The Gini coefficient is the ratio of area between the diagonal and the "Lorenz Curve" and the total area under the diagonal, where the Lorenz Curve traces the cumulative proportion of total income received by different income groups.

[5] If the policy measure is ideal the incidence of the fiscal effects of a tax is just the inverse of the distribution of responsibility for environmental damages.

[6] In some cases, the value of these damages may not be equivalent to the physical level of damages and it is possible that a measure which is progressive in physical terms would be regressive in value terms.

[7] Disaggregation by income class is usually considered an inferior guide to the real relative burden of a tax, since changes in income may reflect demographic (i.e. age), temporary (i.e. unemployment) or exceptional (i.e. income windfalls) factors, whereas expenditure tends to be more stable. Moreover, it is recognised that supplementary forms of disaggregation (i.e. children, pensioners, single parents) can cast further light on the issue, and these will be discussed where relevant.

Table 1

Some Adverse Environmental Effects of Sectors Examined			
	Agriculture	Transport	Residential Energy
Air Pollution	Greenhouse Gases Ammonia	Greenhouse Gases, CO, NOx, PM, Ozone, Lead	Greenhouse Gases SO2
Water Pollution	Nutrients and Pesticides		
Resource Use	Soil, Water and Fossil Fuels	Fossil Fuels	Fossil Fuels
Other	Landscape and Biodiversity	Landscape	

Section II reviews some of the experiences countries have had with ETR. The distribution of fiscal effects is then reviewed in Section III. In Section IV the distribution of some of the most important environmental and social effects is reviewed. Section V will discuss the means whereby the twin objectives of efficiency and equity can be reconciled, and Section VI summarises the arguments presented. These arguments can be summarised as follows:

- In some (but by no means all) cases the distributional consequences of ETR may be distinctly regressive, at least in terms of relative tax burdens.

- The revenue raised by environmental taxes (unlike other environmental policy measures) provide the means whereby some of these adverse distributional consequences can be mitigated and even reversed.

- The distribution of environmental and social consequences are much less readily quantifiable and depend very much on the sector affected and the precise form of the reform introduced.

- In many cases, by removing market failures and barriers which have a particular impact on lower-income households expenditure patterns in environment-intensive sectors, distributional and environmental objectives can be complementary.

On the basis of the evidence reviewed it is concluded that distributional concerns, while important in many cases, should not prevent or delay the introduction of environmental taxes. Rather, they should serve as guiding principles in the design of ETR, not only for their own sake, but also because efficiency objectives and equity objectives can be complementary in a well-designed package of ETR.

II. Environmental Tax Reform: Examples and Extent

To provide a background for recent discussions of environmental tax reform, it is helpful to review the case for economic instruments more generally. Such reforms can take a variety of forms and this section also looks at the various measures which can be classified as environmental taxes. Finally, the section reviews the extent of ETR in OECD economies at present.

II.A The Case for Environmental Taxes

Environmental policy instruments can be broadly divided into two groups: *economic instruments* such as emission taxes and tradeable permits; and *direct regulation* such as technology-based controls and emission standards. As noted above, the perceived superiority of market-based policies relative to direct regulation is a recurrent theme in the literature on environmental policy. This belief rests upon a number of related propositions:[8]

- Since the marginal costs of reducing pollution will vary across sources, controls which mandate equivalent reductions or specific abatement equipment for all emitters will be more costly than a market-based policy which allows individual firms to equate the marginal costs and benefits of reducing emissions.

- It is often argued that monitoring and enforcement costs are lower for economic instruments than for direct regulation. This arises from the fact that the response of firms to a market-based measure will itself elicit information on the costs of abatement and mitigation, facilitating the regulator's role. This is not true of many forms of regulation such as emission controls.

- Direct regulation is purported to be even less cost-effective when dynamic effects are taken into account.[9] While the benefits of technological innovation which mitigate environmental damages do not accrue to innovators under many forms of regulation (i.e. technology-based controls), under a market-based regime the firm itself is able to capture the benefits.

- Finally, given the fiscal constraints faced by many countries, the different effects of alternative instruments on the public purse must also be recognised. A revenue-raising environmental policy instrument - such as taxes or some forms of permit allocation - may be attractive relative to direct controls which often involve increased public spending due to their costs of implementation.[10]

As noted, this report focuses on particular kinds of economic instruments (i.e. taxes). However, as will be discussed below, even within this classification there are a wide variety of instruments which can be applied.[11]

---

[8] See Baumol and Oates (1988), Tietenberg (1990) and OECD (1994a) for discussions and some empirical evidence.

[9] As far back as the mid-1970s it was pointed out that 'over the long haul, perhaps the most important single criterion on which to judge environmental policies is the extent to which they spur new technology towards the efficient conservation of environmental quality.' (Kneese and Schultz 1975).

[10] These costs are also incurred with tax measures, but in most cases they will not exceed the revenue obtained.

[11] This study does not look at the effects of subsidies (i.e. negative taxes). However, many of the arguments presented and evidence provided have direct analogies in cases where existing subsidies are removed.

II.B The Nature of Environmental Tax Reform

Defining environmental tax reform precisely is difficult since the application of such measures is far from straightforward in practice. Many pollution emissions are of the non-point source type (e.g. some agriculture sector effluents). In such cases it is difficult to measure the quantity of pollutant output and design pollutant-based effluent taxes on this basis. In a related vein, monitoring costs of many forms of pollution emission are prohibitively expensive (e.g. some transport sector emissions). In addition, the damages associated with many emissions depend crucially upon the characteristics of the receiving environment (e.g. SO_2 emissions). Finally, many emissions have synergistic effects with other pollutants (e.g. VOCs and NO_x generating ozone), further complicating the application of environmental taxes. Given these and other issues, the application of an 'ideal' environmental tax is an exception, even amongst the sub-set of incentive-based environmental instruments which are applied at present.

Notwithstanding these difficulties, the following list provides a general overview of some of the measures which would tend to be classified as environmental taxes. (See EEA 1996, Gee 1997, OECD 1996, and Gale 1995. Box 1 gives more details on measures applied specifically to the agriculture sector.)

- *Emission and effluent taxes* which are targeted directly at the source of environmental damages. Examples include taxes on ozone-depleting chemicals tax (Austria, Australia, Denmark, USA), nitrogen oxides charges (Sweden), acidifying air emissions tax (France), and water effluent charges (Australia, Canada, Germany, Portugal, Spain, US).

- *Disposal and waste taxes* which are targeted at the unwanted by-products of production processes. Examples include the landfill levy (UK, Denmark, France, Germany, Belgium, Netherlands, Australia, Italy) and charges on hazardous waste (Austria, Belgium, Finland, US).

- *Input taxes* which are targeted at inputs to production which are closely associated with particular environmental damages. Examples include carbon taxes (Netherlands, Sweden, Norway, Denmark), fertiliser charges (Austria, Finland, Norway, Sweden, US), battery charges (Canada, Denmark, Portugal, Sweden), packaging charges (Canada, Denmark, Finland, Norway, Portugal, Sweden, US),), and charges on lubricant oil (Finland, France, Italy, Norway, US).

- *Tax differentiation* and allowances which distinguish between goods or inputs with different environmental impacts. Examples include higher taxes on leaded fuels (most OECD countries), vehicle excise duties adjusted according to vehicle characteristic (Italy, Sweden, Germany and a number of other European countries), and the renewable energy tax credit (US).

- *Charges and fees* which are levied on the use of services with significant environmental impacts. Examples include water charges (most OECD countries), municipal waste user charges (Finland, US), user charges of sewerage (most OECD countries).

Box 1

Environmental Taxes in the Agriculture Sector	
Country	Tax Particulars
Austria	A tax on fertiliser use has been levied since 1986. Current rates are Sch 5 (approximately 24 pence) per kg of nitrogen, Sch 3 (15 pence) per kg of phosphate, and Sch (7 pence) per kg of potassium. The tax revenue in 1990 and 1991 was about Sch 1 billion (approximately £65 million in current terms).
Finland	Tax on phosphate fertilisers levied at 1.5 Markka (approximately 18 pence) from 1991. Since 1988 a tax of 2.5 % is levied on the net selling price of pesticides.
Sweden	Since 1988 a tax on both nitrogen and phosphate fertilisers has been levied. As from 1994, the rate is Skr 1.60 (13 pence) per kg of nitrogen and Skr 30 (24 pence) per gram of cadmium. A pesticide charge is levied at Skr 20 per kg of active substance.
Belgium	A tax on manure surplus to farm requirements has been levied since 1991 by the Flemish region.
Denmark	Retail sale of pesticides sold in containers less than 1 kg or 1 litre is subject to a tax. For imports the rate is 20% of the producer price. Pesticides sold in larger quantities are subject to a tax of 3% of the wholesale price. A new fertiliser tax is proposed for 1999.
Netherlands	A tax is levied on the production of animal manure. The tax rates are based on the weight of phosphate produced on the farm per hectare per year. It is set at the rate of Dfl 0.25 (approximately 8 pence) per kg for farms that produce 125-200kg of phosphate per hectare per year and Dfl 0.5 (15 pence) per kg for farms producing more than 200kg per hectare. Farms that produce less than 125kg of phosphate per hectare per year are exempted from the tax.
Norway	Taxes on fertilisers and pesticides have been in place since 1988. The fertiliser tax is paid by the wholesalers. The rates in 1994 were Nkr 1.21 (approximately 12 pence) per kg content of nitrogen and Nkr 2.30 (approximately 23 pence) per kg content of phosphorus. In 1994 the tax revenue was approximately Nkr 165 million (£16.3 million). The pesticides tax is paid by the pesticides importers. The tax rate is 13% of the wholesale price. In 1994 the tax revenue was approximately Nkr 22 million (£2.2 million).

Given this wide range of measures, the extent to which a particular instrument may be considered an environmental tax is ambiguous.

II.C The Extent of Environmental Tax Reform

Notwithstanding the difficulties arising from definitional issues, efforts have been made to quantify the extent of environmental taxation in OECD countries. According to one review of the origins of government tax receipts, most countries collect between 7% and 9% of their total revenue from 'environmental' taxes (Gee 1997). The European Environment Agency (1996) estimates that the share of taxes on energy and the environment relative to total taxes in the EU in 1993 was 5.2% and 1.5% respectively. The OECD (1996) reviewed the extent of ETR in Denmark, Finland, the Netherlands, Norway and Sweden, finding a range of 5.4% (Finland) to 10.8% (Norway) of total tax revenue.
Given the discrepancies in definitions applied, evidence on changes over time in the extent of environmental taxation may be a more useful indicator than absolute levels. The OECD (1996) recently reported a general increase in the proportion of government revenue raised by 'environmental' taxes in three countries (Netherlands, Norway and Sweden) over the period 1980-1993, while trends elsewhere were approximately constant (Denmark) or showed an appreciable decline (Finland). Over the same period, in the EU as a whole (EEA 1996), there has been a slight increase in the relative tax burden on energy, but not on the environment.

Thus it appears that the proportion of the total tax burden which falls on the environment is modest even in those countries which have been leaders in the introduction of environmental measures. One reason for the apparent reluctance of governments to go further may have been concerns about the impact of environmental policies on competitiveness. (See Barker and Johnstone 1998 for a discussion of carbon taxation and international competitiveness.) Nevertheless, the burden of tax on the environment does seem to have increased in recent years and is likely to continue to increase in future.

III. The Distribution of the Financial Effects of Environmental Taxation

A key issue in assessing the impact of environmental tax reform is how it affects different groups in society. Looking first at financial effects, one can distinguish three avenues by which environmental taxes will affect different groups: the direct and indirect effects of environmental taxes on consumer goods markets; the effects of behavioural responses to price changes; and, the effects of recycling the revenue generated. These different impacts are illustrated here by a review of taxes on petrol, domestic fuel, carbon/energy, and agricultural inputs.

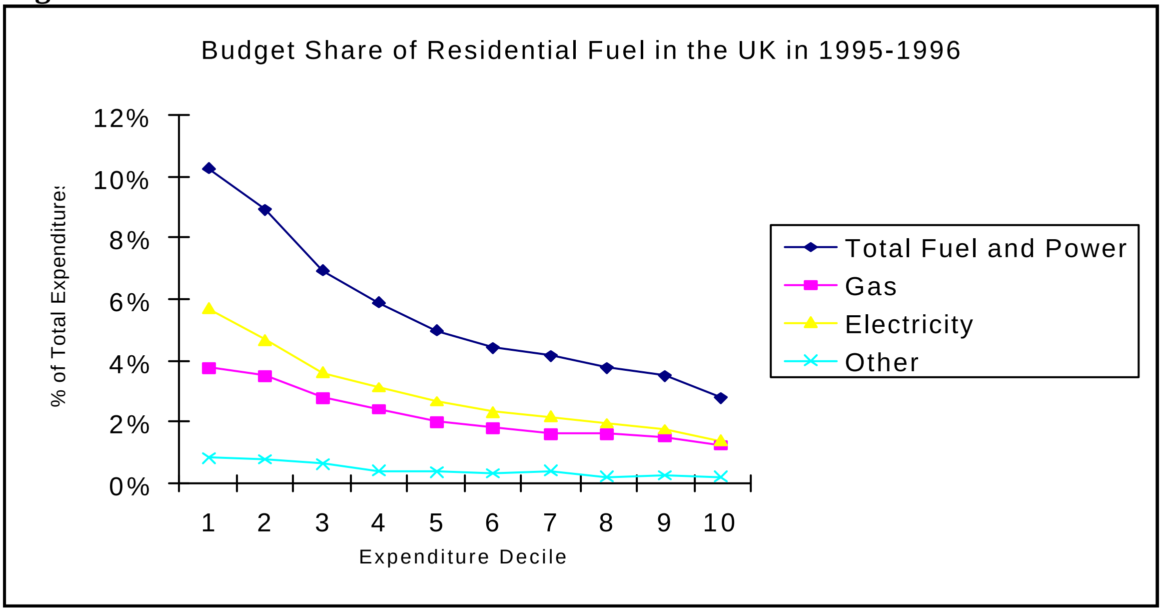
III.A Expenditures on Environmentally-Sensitive Goods by Income Group

Widespread concern about the potentially regressive impact of many environmental taxes arises largely from the observation that lower-income households tend to devote a larger proportion of their household budget to those goods and services which are directly affected by environmental tax reform. The evidence for this claim will be examined for the three sectors (energy, transport, and agriculture) in turn.

Spending on Energy

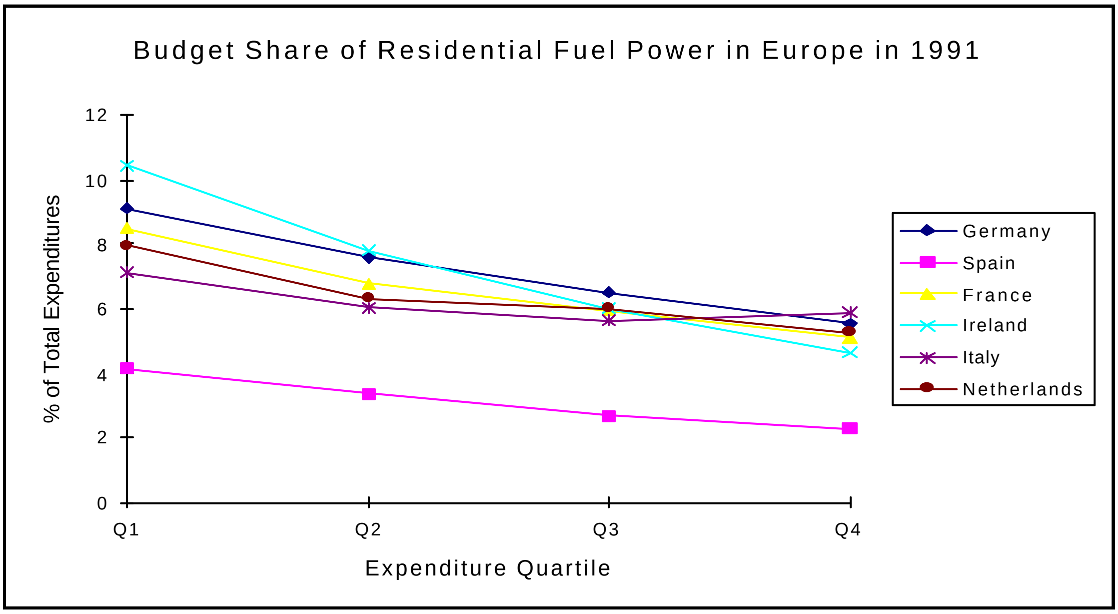
Using expenditure deciles as the means of comparison (i.e. dividing the total population into ten segments of equal size grouped by expenditure levels), Figure 1 shows the share of total spending on residential fuel and power by expenditure decile in the UK in 1995-1996 (UK ONS 1996). This shows an appreciable decline in the share of gas, electricity and 'other' (i.e. solid and oil) energy carriers in total expenditure as average spending power increases. Thus, while the lowest decile devotes over 10% of total spending to fuel and power, the richest decile devotes less than 4% to the same purpose.

Figure 1



This pattern appears to hold elsewhere. For instance, Poterba (1991) found that in the mid-1980s 7.6% of total expenditures for the lowest expenditure decile of American households went on fuel while the equivalent figure for the highest decile was 4.0%. Similarly, Smith (1992) found that in five of six EU countries examined in 1991 (Spain, Italy, Netherlands, France and Germany) domestic fuel expenditures as a proportion of total expenditures in the lowest quartile were 2-3% higher than for the highest quartile. Ireland exhibited an even more significant decline in the share of spending on domestic fuel in total household expenditures, from almost 11% of total expenditure for the lowest quartile to approximately 5% for the highest quartile. (See Figure 2.)

Figure 2



In this context, it is not surprising that one of the most controversial 'environmental' taxes was a proposal by the last Conservative (UK) government to raise VAT on domestic fuel. Indeed, it was largely on social and distributional grounds that the tax was ultimately defeated in Parliament.[12] However, prior to its defeat Crawford, Webb and Smith (1993) estimated the potential effects of the introduction of 17.5% VAT on domestic fuel in the UK. They found that the lowest income quintile would have faced additional indirect tax payments equal to 2.0% of total expenditures, while the richest quintile would have paid an additional 0.6% of their total expenditures on higher tax. Pensioners would have been hit particularly hard. The lowest income quintile of pensioners would have paid over 50% more than non-pensioners in the lowest-income quintile.

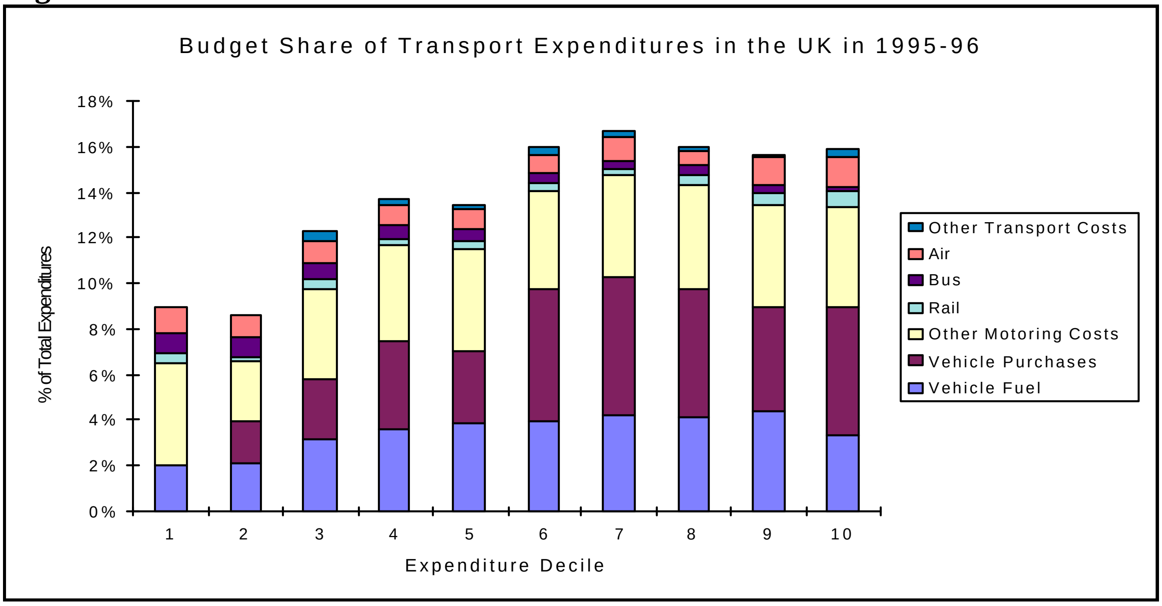
Moreover, in the event that taxes on domestic fuel were targeted directly at the environmental characteristics of the fuel, the effects would tend to be even more regressive, as poorer households tend to use fuels which are more carbon- and, to some extent, sulphur-intensive. In the 1980s in the UK, 77% of households in the richest income quartile used the 'cleanest' fuel (gas) as their primary heating source, while only 63% of the poorest quartile did so. Conversely, 15% of the poorest quartile used the 'dirtiest' fuel (coal) and only 9% of the richest did so (UK DOE 1991).

Spending on Transport

[12] And, in fact, subsequently lowered by the incoming Labour government to 5%, which was well below the average rate on other goods and services.

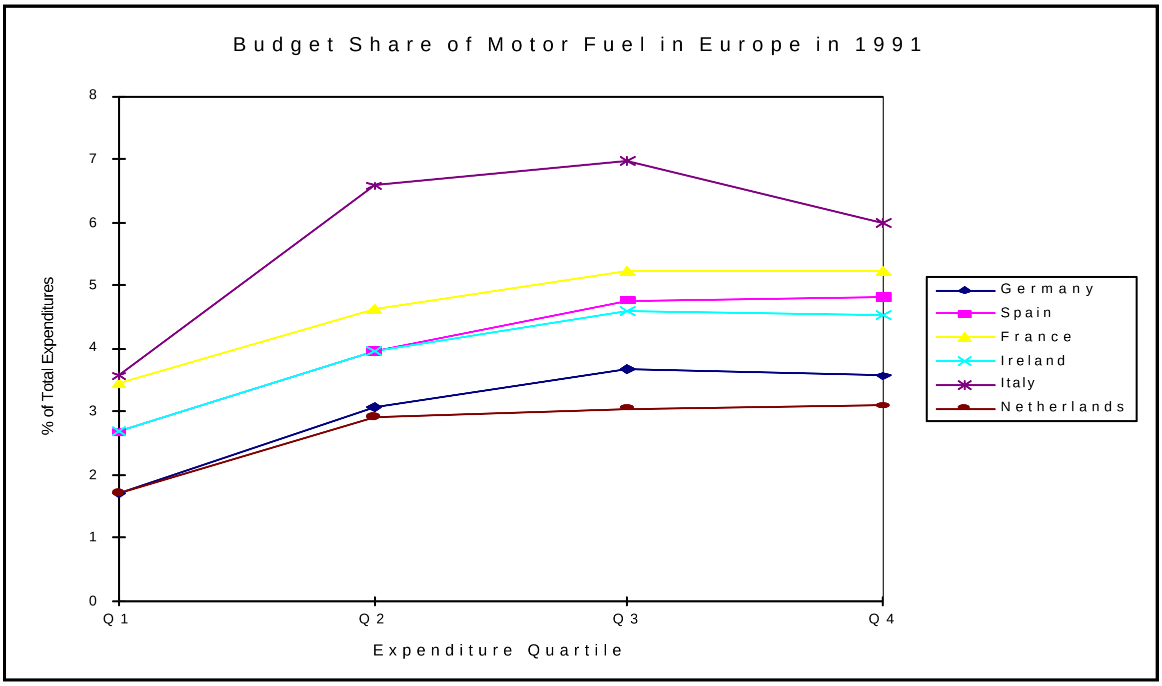
The pattern of transport expenditures is rather different. Once again using UK data, Figure 3 shows the generally rising trend in proportional expenditure per decile, as total expenditures increase. Most significantly in environmental terms, the share of fuel expenditures rises from 2.0% to 4.3% for the second highest decile, before falling to 3.3% for the richest decile. Taking fuel and vehicle purchases together, the proportion rises until the seventh decile and then falls for the last three. Thus, taxes on vehicles and/or fuels appear to be relatively progressive. Expenditures on bus and rail are approximately proportional by expenditure decile (between 1.0% and 1.4%).

Figure 3



The breakdown of petrol expenditures in other countries shows the same general tendency as in the UK. For instance, Casler and Rafiqui (1993) find that in the United States gasoline expenditures rise from 4.5 % of total expenditures for the lowest quintile to 5.1% for the fourth quintile, before falling appreciably to 3.9 % for the richest quintile. In continental Europe in 1991 (Figure 4) relative expenditures on motor fuels of the poorest quartile are less than those of the other quartiles in all six of the countries examined (Smith 1992). The average ratio of proportional expenditures by the richest and lowest quartiles is 1.73.

Figure 4



Prior to many of the recent increases in petrol duties, Johnson, McKay and Smith (1990) reviewed the effects of the imposition of a tax of 55 pence per litre on petrol. They found that the effects were approximately proportional across income levels for car-owning households, but distinctly progressive across all households. Similarly, taxes on vehicles themselves are also not thought to have distributionally adverse consequences.

This result is hardly surprising given that the rate of car ownership among households in the lowest income decile was 18.2%, while the richest decile had a car ownership rate of 97.5%. Moreover, over 70% of the latter had more than one vehicle (UK ONS 1996). Thus, a tax on vehicles is likely to have similarly progressive distributional consequences, although this will depend upon the relative intensity of car use for different socio-economic groups and the relative frequency of vehicle purchases. Other vehicle-related environmental measures which are more closely related to emission levels have more ambiguous effects. For instance, Ferguson and Taylor (1996) review the effects of 'Green' vehicle excise duties in Europe and find that a scheme based on engine size is likely to be progressive, while one which is based on vehicle age is likely to be regressive (at least for car-owning households). This finding is a reflection of the characteristics of the cars owned by different income groups, ie richer households tend to buy new cars.

Spending on Agricultural Products

It has long been observed that poor households spend a higher proportion of their income on food than rich households do. The percentage expenditures on food by income groups for the UK, US and Canada are shown in the Table 2. For the poorest 10% in the UK, food and non-alcoholic drinks account for 25.2 % of total weekly expenditures. For the richest 10%, on the other hand, these items account for only for 15.1% of weekly expenditures. Similarly, households with lower incomes in both the US and Canada will spend a greater proportion of their income on food.

Table 2

Relative Household Expenditures on Food in the UK, the US and Canada					
UK Income Decile	Household Share (%) of Food Expenditures	US Income Bracket (US$000s)	Household Share (%) of Food Expenditures	Canadian Income Quintile	Household Share (%) of Food Expenditures
Decile 1	25.2	4.9 or less	36.9	Quintile 1	19.1
Decile 2	24.6	5 - 9.9	34.8	Quintile 2	16.0
Decile 3	22.1	10 - 19.9	24.2	Quintile 3	13.7
Decile 4	20.8	20 - 29.9	21.0	Quintile 4	12.1
Decile 5	19.7	30 - 39.9	17.3	Quintile 5	10.1
Decile 6	18.3	40 - 49.9	13.6		
Decile 7	18.5	50 - 69.9	11.9		
Decile 8	17.2	70 over	8.5		
Decile 9	16.9				
Decile 10	15.1				
Sources: UK ONS (1998), US DA (1996), StatsCan (1992).					

It is generally assumed that environmental tax reform in the agriculture sector would entail a rise in food prices, due to increased production costs. Simulating the effects of a 5% rise in the price of all foods in the UK, Johnson, McKay and Smith (1990) find that the tax burden is somewhat regressive, with the poorest decile paying 0.31% of their total expenditures in increased tax payments, while all other deciles paying between 0.26% and 0.28%. Not surprisingly, households with no children are less adversely affected than households with children.

However, it would be exceedingly incautious to use such a policy scenario to evaluate the distributional consequences of ETR in the agriculture sector, since most measures are likely to be targeted at specific environmental impacts (i.e. from chemical pesticide or fertiliser use, or from water consumption) which may differ widely by food type. For instance, comparing recorded sectoral pollution “incidents”[13] in the UK with relative expenditures by the bottom and top income deciles confirms that measures targeted more closely at the “environment-intensity” of different crops would be regressive.[14] (See Table 3.) Indeed, relative to an environmental measure which targeted all foods equally, it would appear to be even more regressive, with lower-income households spending a relatively higher proportion than average in those food groups responsible for pollution incidents.

[13] For non-point sources such as emissions and effluent from the agriculture sector it is common to use “incidents” (discrete events) as a measure of pollution-intensity rather than emission rates, due to the difficulties in monitoring pollutants directly.

[14] Arable crops and mixed crops were responsible for 6% and 4% of “incidents” respectively but it is difficult to trace them directly to different food groups.

Further light can be cast on this issue by hypothesising a measure which increased the cost of pesticide application. In the UK, fruits tend to be the most pesticide-intensive crops, often requiring as many as 20 separate applications (WWF/CPRE 1996). In 1995-1996 fruits represented 5.96% of total food expenditures of the lowest-income decile in the UK, and 7.31% for the highest-income decile, indicating significant progressivity. However, as a proportion of total expenditures, the figure is 1.25% for the lowest decile and 0.66% for the highest decile (UK ONS 1996). Thus, even if the environmental measure introduced hits foods to which the rich devote a greater share of their food budget, there is still a considerable degree of regressivity in terms of the impact on total household spending.

Table 3

Pollution "Incidents" and Expenditure Patterns in the UK Agriculture Sector							
		Expenditures (£/HH/Week)		As % of Food Expenditures		As % of Total Expenditures	
	% of Sectoral Incidents	D1	D10	D1	D10	D1	D10
Dairy	55%	3.08	8.60	16.68%	15.18%	3.51%	1.38%
Pig	7%	0.75	1.97	4.06%	3.48%	0.85%	0.32%
Poultry	2%	0.74	2.89	4.01%	5.10%	0.84%	0.46%
Sheep	2%	0.31	0.91	1.68%	1.61%	0.35%	0.15%
Source: "Incidents" from WWF-UK/CPRE 1996 and Expenditures from UK ONS (1996)							

Moreover, poorer households tend to consume rather more of those crops which are not particularly water-intensive (i.e. potatoes) and less of those which are (i.e. fruit). For instance, Tsur (1993) found that the water-intensity (irrigation only) of potatoes was 75% that of vegetables and only 18% that of citrus fruits. Comparing this with expenditure patterns by food type suggests that ETR in the agriculture sector which increased the price of water for farmers would not necessarily be as regressive as is often assumed.

Indirect Expenditures

Imputing the relative incidence of ETR on different groups solely from such direct expenditure shares would be incautious. Attention must also be paid to the environment-intensity of inputs used in the production of goods and services which are affected by any taxes introduced. This is particularly the case for energy taxes, which not only affect direct expenditures on fuel, but also non-fuel consumption patterns as well as the energy-intensity of inputs used in the production of all goods. This in turn reflects the importance of energy in all sectors of the economy.

The importance of these indirect impacts is illustrated in Table 4, which presents the carbon-intensity of commodity groups in the UK disaggregated in three senses:

- Direct Consumption - e.g. emissions arising from the direct consumption of fuels used by household appliances or vehicles;

- Direct Production - e.g. emissions arising from the manufacture of vehicles and appliances; and,

- Indirect Production - e.g. emissions arising from the manufacture of steel used in the production of vehicles and appliances.

Table 4

Estimated Carbon-Intensity of Selected Commodities (CO_2 kgs / £) in the UK				
	Direct Consumption	Direct Production	Indirect Production	Total
Household Energy	5.57	5.61	0.98	12.15
Petrol	5.05	0.23	0.06	5.33
Personal Services	0.00	0.25	2.24	2.49
Vehicle Repairs	0.00	0.22	1.89	2.11
China & Glassware	0.00	0.58	0.72	1.31
Public Transport	0.00	1.12	0.15	1.27
Taxis & Car Hire	0.00	0.72	0.28	1.00
Source: Symons, Proops and Gay (1994).				

While fuels remain the most carbon-intensive commodities, other goods (i.e. china and glassware) and services (i.e. personal services), which have no direct consumption-related emissions, nevertheless exhibit very high emissions in production (direct and indirect). Significantly, public transport is quite high (although low relative to private motoring), revealing the environment-intensity of mobility generally.

Taking these factors into account, the distributional effects of an energy tax are quite different than would appear to be the case from a comparison of direct consumption expenditures. For instance, Casler and Rafiqui (1993) compare the direct and indirect effects of an energy tax in the United States. While a tax on final consumption would have significantly regressive effects, this is less true if indirect effects are considered also. Thus, proportional direct expenditures on fuel are 26% lower for the highest expenditure quintile relative to the lowest quintile, but only 6% lower when indirect expenditure is included.

Despite the mitigating effects of including indirect expenditures on energy, it is generally found that energy taxes remain somewhat regressive. Poterba (1991) examined the effects of a $100 per ton carbon tax in the United States and found that the incidence of the tax was equal to an average of 3.7% of total expenditures for the three lowest deciles, but only 2.6% for the three highest quintiles.[15] Symons, Proops and Gay (1994) examine the effects of a carbon tax sufficient to reduce total UK emissions by 20% (£240.50), and find that the Gini Coefficient increases from 0.386 to 0.397. Cornwell and Creedy (1996) conduct a similar exercise for Australia and find that effects are also regressive, but slightly less so than in the case of the UK.

[15] The results are even more startling relative to income (i.e. 10.0% and 3.7%).

One of the few studies to attempt to determine the full direct and indirect effects of more general environmental policies is Robison (1985), which looks at the relative cost burdens of American pollution control policies. He finds that low-income households paid as much 1.09% of their income on industrial pollution abatement (directly and indirectly), while high income households paid as little as 0.218%. In a somewhat dated study Gianessi *et al* (1979) reached similar conclusions. However, in both studies the policies examined included both taxes and direct controls.

III.B Behavioural Responses to Price Changes by Income Group

While the level of expenditures on those goods and services whose prices rise following the introduction of environmental taxes is clearly the most important determinant of the distributional effects of the reform, it is also important to examine how households respond to changes in prices. Relative pre-tax expenditure levels (direct and indirect) will tend to overstate the financial burden for all households since they will tend to substitute away from goods whose prices have increased. The more formal studies cited above tend to include behavioural responses by households to changing prices. This is important since if demand is less elastic (i.e. households are less responsive) for those goods which are consumed in greater proportion by lower-income households, then the tax will be more regressive (in financial terms) than in cases where households respond more elastically.

This issue can be explored in greater detail with reference to agriculture. Recent American estimates reported in Table 5 from Huang (1996) indicate that the price elasticities (% change in expenditures for a percentage change in price) of various food categories are less than one. This suggests that food consumption is not particularly responsive to changes in the prices of food products. However, in the event that ETR in the agriculture sector affects different food items differentially - i.e. pesticide-intensive crops like fruits and vegetables - then there may be some substitution possibilities between food groups. (This issue is explored in greater detail in Section IV.B.)

Table 5

Own-Price Elasticities Of Selected Food Items in the US	
Food Category	Price Elasticity
1. Beef	-0.62*
2. Pork	-0.73*
3. Chicken	-0.37*
4. Eggs	-0.11*
5. Milk	-0.04
6. Flour	-0.08
7. Potato	-0.10*
8. Apple	-0.19
9. Orange	-0.85*
10. Banana	-0.50*
11. Lettuce	-0.09
12. Tomato	-0.62*
13. Onion	-0.21*
14. Carrot	-0.53*
15. Peanut	-0.17*
16. Coffee	-0.18*
Source: Huang (1996). * indicates statistical significance at least at the 5% confidence level.	

Another important factor not usually addressed in the empirical studies is difference in the degree of substitution possibilities which exists for particular environment-intensive goods by income group. In the event that the price elasticity of demand for environment-intensive goods is lower for low-income households, then the regressive distributional effects of the tax will be even more pronounced than has usually been estimated. In general, there are good reasons to believe that lower-income households will tend to be more constrained in their choices for certain goods (and thus have lower price elasticities) since they face more obstacles in the marketplace.

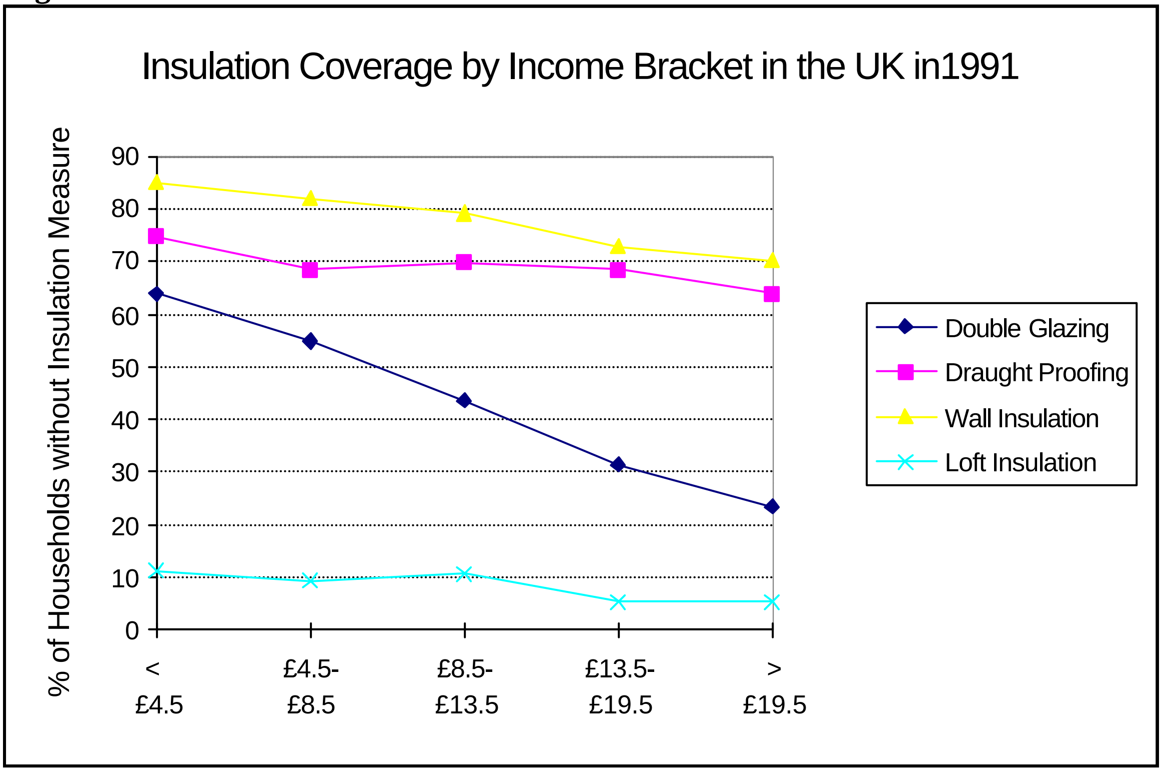
This can be illustrated with reference to demand for household fuels. Firstly, it is important to recognise that fuels are not demanded in and for themselves, but rather as a means of providing an energy service. For instance, in the case of space heating it is dwelling warmth which is the energy service, and it is produced jointly by the fuel, the heating appliance, and the thermal characteristics of the dwelling. Thus, in the face of changing absolute and relative fuel prices the household's reaction is not reflected purely in terms of the quantity of fuel consumed since there are other means by which a household can achieve a given level of warmth. However, the structure of the market is such that high-income households may be better able to substitute other inputs (i.e. change their appliances, substitute other fuels, or increase insulation levels) in the face of rising fuel prices. In such cases the price-elasticity of demand for fuel used in heating the home would tend to be higher for them than it is for lower-income households.

There are many reasons why this may be so due to the existence of market failures or barriers which particularly affect lower-income households.[16] On the one hand, since the

[16]See Barker and Johnstone (1993), and Brechling and Smith (1994) for discussions.

initial costs of reducing heat loss or switching fuels can be quite significant, low-income households are less likely to undertake such investments because of inadequate access to savings and/or credit.[17] Indeed, even at prevailing fuel prices capital constraints appear to be significant, despite the high returns which exist for many measures. For instance, the return on investment for loft insulation is often in excess of 30%. (See Barker and Johnstone 1993.) However, in 1991 many low-income households had not invested in insulation. (See Figure 5 which gives insulation ownership by income bracket in £1000s.)[18] This can certainly be attributed in part to the fact that access to loans is often restricted for low-income households relative to the credit risk they actually pose.

Figure 5



On the other hand, given that low-income households are much more likely to be tenants than high-income households, their incentive to undertake such investment is correspondingly lower. Whereas 20.4% of households in the UK in the lowest total expenditure bracket (less than £100/week) were owner-occupiers in 1991, the figure for the highest income bracket (more than £550/week) was 93.3% (CSO 1992). Since landlords are usually responsible for capital costs and tenants are responsible for running costs, neither party is

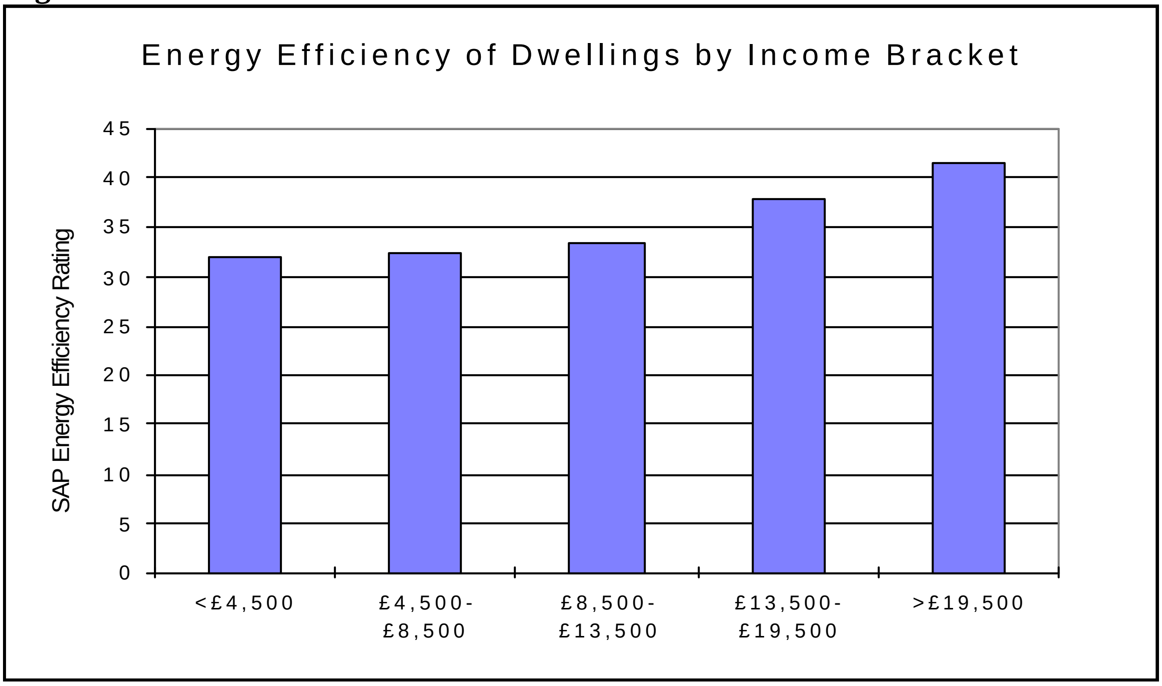
[17] It has been estimated that the cost of self-installation of cavity wall insulation and loft insulation is about £300-£500 and £110-£160 respectively for a terraced house. Double glazing may cost anywhere from £120-£600 depending on the number of windows, while draught-proofing may cost as much as £60. Moreover, the cost when such work is undertaken by professional contractors can be as much as twice that of self-installation (UK DOE 1993). The cost of installing an efficient gas-fired condensing boiler for a small dwelling was estimated to be between £400 and £600 more than a standard central heating boiler (UK DOE 1993). An efficient off-peak electric storage boiler is estimated to cost between £2,500 and £3,000 (UK EEO 1990). These figures are clearly beyond the means of most low-income households.
[18] Note that the figure for loft insulation only includes households with lofts.

likely to have sufficient incentive to invest in home heat retention improvements and fuel substitution which involve high capital costs but result in lower running costs.[19]

Using the Standard Assessment Procedure of energy efficiency rating,[20] the DOE's *English House Conditions Survey*, reports that energy efficiency for owner-occupiers was markedly higher (SAP rating 37.2) than private tenants (SAP rating 22.4) in 1991 (UK DOE 1996). In a formal study, Brechling and Smith (1994) find that investments in loft insulation, wall insulation and double glazing are significantly related to housing tenure, with owner-occupiers being much more likely to undertake such investment. While capital constraints may also be important - i.e. income levels are positively and significantly related to the probability of undertaking such investments - the relationship is not as strong and there are a number of alternative hypotheses which would support such a relationship.

Thus, in general, it would appear that there is a positive relationship between the heat retention characteristics of housing and income levels. The DOE's *English House Condition Survey* found that lower-income households were much more likely to live in energy inefficient dwellings (UK DOE 1996. See Figure 6). This relationship between energy efficiency and income/socio-economic status has significant consequences in terms of the distribution (and, more generally, welfare) effects of a carbon/energy tax, since relatively more fuel is required to achieve a given level of warmth for low-income households. (This issue is explored in Section IV.B below.)

Figure 6



[19] A third possible market failure which is frequently cited is related to information. However, its effects on household energy systems do not appear to be systematically related to income levels.
[20] The SAP rating is a logarithmic scale which varies from 1 (highly inefficient) to 100 (highly efficient), and measures both the rate of heat loss (principally thermal efficiency) and the cost of supplying the heat (principally conversion efficiency).

While the discussion in this section has focused on taxation of residential energy, the general argument is true for a wide variety of goods which are likely to be affected by environmental tax reform. For instance, low-income households which own cars may be less able to substitute their car for a more efficient model in the face of rising fuel prices. However, since the ratio of transport fuel costs to vehicle costs is much lower than the ratio of domestic fuel costs to insulation or boiler costs, this problem is likely to be much less acute. Moreover, since low-income households often do not own cars, this issue will only affect a subset of households, such as those who do not have access to adequate public transport in rural and suburban areas.

More importantly, emphasis on these short-term effects is somewhat misguided in a sector such as transport where the infrastructure is very long-lasting. In the longer run it may well be that measures such as petrol taxes increase mobility, even for car-owning households. There are two reasons why this is so: first, measures such as petrol taxes are likely to make public transport projects more viable in a financial sense as passengers substitute away from car use; and, second, it is quite likely that any ETR which significantly increases petrol prices will be accompanied by more far-reaching policy changes which encourage investment in public transport options. Therefore, access to transport options for some low-income households may actually rise following the introduction of ETR in the sector.

III.C Revenue Recycling and Mitigating Policies

As noted above, in most discussions of the effects of introducing environmental taxes it is implicitly assumed that the tax is not of sufficient economic importance to warrant a broader package of fiscal reform. However, given the magnitude of revenue raised under many of the environmental taxes discussed, this assumption is both unnecessarily restrictive and politically implausible. Moreover, since the means by which the revenue is recycled may have at least as significant distributional consequences as the tax itself, in this section these issues will be explored.

The potential for environmental taxes to raise revenue can be illustrated from some of the discussions of the potential fiscal repercussions of a carbon tax. For instance, using a macroeconomic model of the UK economy (Barker, Baylis and Madsen 1993) it has been estimated that the proposed CEC tax would have raised approximately 5 billion in revenue in 1995 rising to 12 billion in 2000. Pearson and Smith (1991) estimated that the tax would have raised approximately 3% of EU governments' total tax receipts, with the proportions being greater than 4% for some of the member countries (Greece and Luxembourg). Symons, Proops and Gay (1994) estimate that a carbon tax sufficient to reduce UK CO_2 emissions by 20% (£240) would increase revenue by 47.7% if there were no other policy changes. Finally, Poterba (1993) estimates that a carbon tax of $100/ton would generate revenue in the region of $200 billion (3% of GNP) in the United States.

Given the pervasiveness of energy use in industrialised economies, it is not surprising that carbon and energy taxes raise significant amounts of revenue. However, even rather specific environmental taxes can raise significant amounts of revenue. In 1994 the American 'ozone depleting chemicals tax' raised $1 billion. The 'water pollution levy' in the Netherlands raised over $750 million in 1993. Prior to its introduction it was estimated that the UK's 'landfill levy' would generate $280 million. And finally, Sweden's 'nitrogen oxides charge' raised $72 million in 1992. (Figures obtained from Gale et al 1995.)

In light of these revenue flows, it is likely that widespread environmental tax reform will be introduced with a corresponding decrease in other direct or indirect taxes, and/or an increase in public expenditures. Indeed, it has been argued that under certain circumstances such a reform might even generate a 'double dividend' whereby the tax reform would reduce both environmental and economic distortions. (For a review of the conditions under which this is likely to take place see Goulder 1994. Other discussions include Parry 1995, and Bohm 1997.) While there have been some debates about the relative likelihood of revenue recycling resulting in such a 'double dividend', there is no question that the form in which the proceeds from taxes are used will have significant distributional consequences. Whether or not the combined effect is progressive or regressive will depend upon the specific characteristics of the fiscal reform proposed and the structure of the economy. In general, recycling can take one of three forms: automatic recycling, reducing other taxes, and increasing expenditures.

Automatic recycling arises from the effect of increased prices for consumer goods and services on existing benefit schemes, state pensions and other indexed receipts. (See Poterba 1991 and Crawford, Smith and Webb 1993.) To the extent that households will differ in terms of the proportion of their income which is indexed, automatic recycling will have distributional consequences. Moreover, since poorer households tend to receive a higher proportion of their income as indexed benefits, the effects are likely to be progressive. However, there are three qualifications which must be made to this argument. First, since the aggregate price indices which are used to calculate benefits often assume a lower proportion of expenditures on necessities than is the case for low-income households, they will not be compensated fully for environmental taxes which hit necessities (i.e. fuel and food) particularly hard. Second, since there is great diversity amongst low-income households, some households (i.e. those which are not eligible for means-tested benefits) will face particular hardship. And, third, since take-up for most schemes is less than 100%, coverage will be less than universal, even amongst eligible households.

The importance of the appropriate indexing of benefits can be illustrated with reference to the treatment of food costs in the determination of social security payments in the UK. The system is based upon a subsistence scale originally derived from poverty studies conducted in the 1930s (NCC 1995). Conversely, several European countries (e.g. Sweden, Netherlands, Norway, Denmark and Ireland) have developed budget standards in which food and other basic necessities are explicitly taken account of in the determination of state benefits. However, of particular relevance to this study is the use of an uprating formula in the UK system which is based on an inappropriate price index.[21] In such a system the amount of revenue recycled automatically to lower-income households following the introduction of ETR in the agricultural sector would be quite different than in continental European countries.

[21] Although an earnings index (which tends to rise faster than price indices in a growing economy) was used in the period 1975-1980.

The distributional effects of recycling via reduced taxes and/or increased discretionary expenditures are more complicated. In general, indirect taxes tend to be regressive, while direct taxes and discretionary expenditures tend to be progressive. Therefore, a reform whereby the proceeds are recycled via reduced income taxes will tend to have more adverse (or less beneficial) distributional effects as one whereby the proceeds are recycled via reduced expenditures on discretionary benefits or reduced value-added-tax. However, this will very much depend upon the precise nature of the fiscal reform. For instance, the revenue may be recycled by increasing tax allowances or marginal rates for the lowest income band. Similarly, it may be recycled by decreasing VAT on goods consumed disproportionately by poorer households.

Symons, Proops and Gay (1994) estimate the effects in the UK of recycling the revenue generated by a carbon tax in a number of manners, including reducing or abolishing VAT, reducing petrol excise duties, and increasing benefits (minimum income provision, increasing pensions, and increasing child allowances). The reform which comes closest to meeting environmental objectives (approximately 20% reduction in CO_2), distributional objectives (not significantly exacerbating inequality) and fiscal objectives (approximate revenue neutrality) is one which involves a 5% rate on VAT, a 30% reduction in petrol excise duties, a minimum expenditure allowance of £45, an increase in pensions of £15, and an increased child allowance of £15.[22]

Cornwell and Creedy (1996) look at the effects of recycling the revenue generated by the imposition of a carbon tax in Australia through adjustments in the minimum income guarantee. While the direct effect of the tax is regressive, the revenue generated is sufficient (an increase in the tax ratio from 0.33 to 0.40) to allow for the reversal of the adverse effects of the tax on income inequality by raising the minimum income guarantee by 50%. They find similar responses for fuel taxes and food taxes, although in these two cases they do not include behavioural responses to the price changes.

Other studies have also looked at recycling the revenue from more specific taxes. For instance, Crawford, Smith and Webb (1993) look at the distributional effects of recycling the revenue from the imposition of 17.5% VAT on domestic energy in the UK through a 0.4% increase in all benefits on the one hand and through an equal-cost increase in means-tested benefits on the other hand. Not surprisingly, both measures are progressive, but the latter is significantly more so.

However, not all means of recycling are likely to be progressive. In the case of California, Walls and Hanson (1996) look at the distributional effects of recycling the revenue generated by the imposition of vehicle registration fees based on environmental characteristics (i.e. estimated emissions). They find that by using the revenue to abolish the existing vehicle registration fee system, which is based upon monetary values, the tax change

[22] The tax rate, however, is much higher (£444), than in reforms which ignore distributional concerns (£277). See the next sub-section for a discussion of this issue. Other means of recycling the revenue either result in large increases in inequality (zero-rating VAT and halving the petrol excise duty) or do not meet the environmental target (halving petrol excise duty and increasing benefits).

is likely to have adverse distributional consequences due to the preponderance of less expensive but older and more polluting cars amongst lower-income households. In this case the 'environmental' tax is more regressive than the 'value' tax which it replaces.

III.D Conclusion

In at least two of the cases examined (energy and agriculture) ETR is likely to have adverse distributional consequences, at least in terms of direct tax burdens. This arises from the fact that domestic energy consumption and food are basic needs, consumed in relatively greater proportion by lower-income households. In the case of transport ETR is unlikely to have regressive consequences in terms of tax burdens. This is due to the higher levels of car ownership for richer households. Moreover, even for those measures which are regressive in terms of direct expenditures, the inclusion of indirect expenditures tends to mitigate the adverse distributional consequences somewhat.

Conversely, in some cases the adverse distributional consequences of regressive environmental taxes may be even more unfavourable than is often presumed when taking into account that poorer households are less able to adjust their consumption patterns in the face of changing prices. However, it has also been shown that since environmental taxes (unlike other environmental policy instruments) raise significant amounts of revenue, some of the adverse distributional consequences of ETR can be overcome by lowering other taxes or increasing expenditures.

IV. The Distribution of the Environmental and Social Effects of Environmental Tax Reform

In this section the distribution of environmental and social effects from ETR will be discussed. In both cases the effects are more difficult to quantify since the consequences are usually realised in non-monetary form and/or are the outcome of complex interactions in the economy. Nonetheless, some general insights can be gained from a close look at the literature.

IV.A Distribution of Environmental Effects of ETR

Just as it is argued that the costs of environmental tax reform tend to be borne disproportionately by the poor, it is also often argued that the benefits are realised disproportionately by the rich. (See Baumol and Oates 1988.) Part of the reasoning behind this assertion arises from a belief in the 'income-elasticity' of demand for environmental quality. Thus, it has often been argued that preferences for environmental quality are income-elastic (i.e. demand for environmental quality rises more than proportionately with income levels). According to this view, environmental tax reform implies targeting basic needs in order to satisfy luxury demands.[23]

---

[23] To some extent the environmental effects of ETR are just the inverse of the fiscal effects. If households respond elastically to changing prices for affected goods then the revenue raised will be quite small but the environmental benefits will be quite high, and vice versa. Thus, for a given tax rate, the elasticity will, in effect, determine the split between fiscal and environmental effects.

As we have seen, the case for the cost side of this assertion is more complicated when the full effects of different types of fiscal reform are analysed. Given the wide variety of attributes and resources associated with environmental 'quality' - i.e. from the health effects of urban air pollution to the aesthetic value of pristine landscapes - the benefits side of such an assertion is meaningless at this degree of aggregation. Moreover, there is increasing reason to believe that there is not a positive relationship between income levels and proportional demand for a number of environmental attributes and resources for which one would imagine such a pattern to exist.[24]

More importantly, there is also good reason to believe that actual levels of exposure to environmental degradation are not uniform across different household groups. If this is the case then households will benefit differentially from environmental tax reforms which reduce environmental damages. In particular, households which are most exposed to pollution emissions will benefit disproportionately from reduced emissions, since prior to their reduction they will have borne a disproportionate share of the costs from the emissions. There are at least two reasons to believe that it is the poor who are most likely to face significant levels of exposure. Firstly, to the extent that property prices rise with income levels, wealthier households appear to be willing to 'insulate' themselves from environmental degradation. Secondly, higher income levels do give some households the opportunity to 'buy' environmental quality (i.e. through holidays, private gardens or second homes) even when their local environment is degraded.

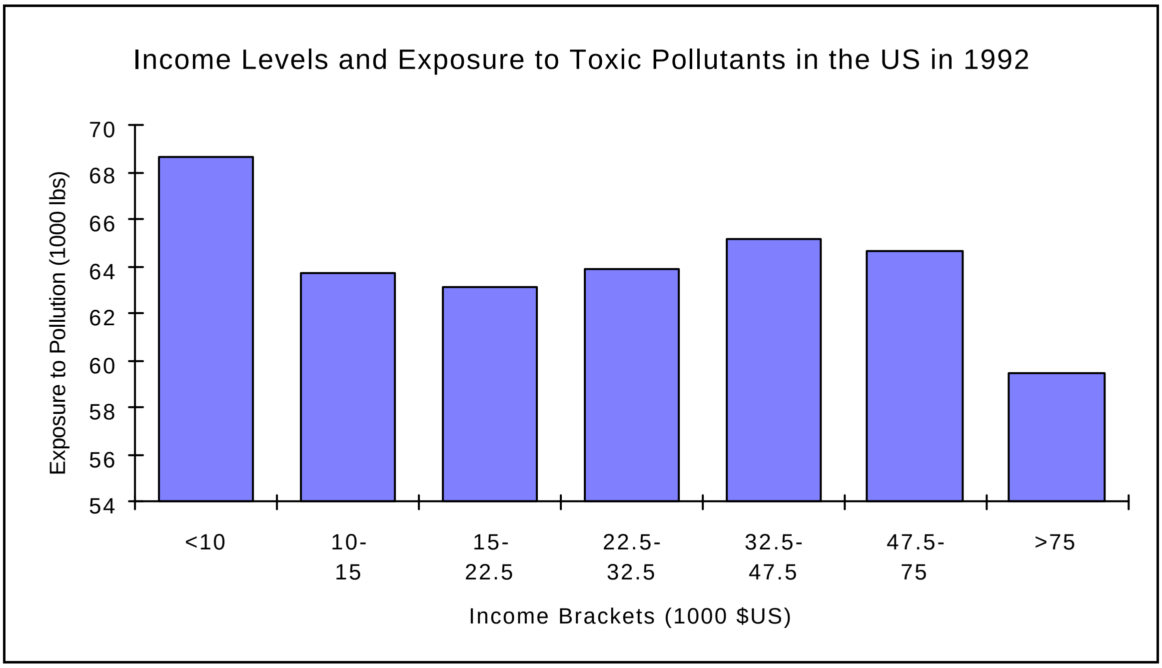
Exposure to Toxic Pollutants

Regrettably, due to data constraints there is very little empirical work on this question in the case study sectors chosen. However, in recent years there has been a spate of studies in the United States examining the relationship between the spatial distribution of toxic releases from industrial plants - usually using the US EPA's 'Toxic Releases Inventory' - and the socio-economic and demographic characteristics of surrounding neighbourhoods. (See Arora and Cason 1996 and Brooks and Seith 1997 for two examples.) In general, the studies find that the lowest income group has the highest degree of exposure and the highest income group has the lowest degree of exposure. (See Figure 7 for some results from Brooks and Seith 1997.) For instance, those below the officially-defined poverty line in the US faced exposure levels 16% higher than those above the poverty line. Not surprisingly, the studies also find that urban households are more exposed than rural households.[25]

Figure 7

[24] For instance, in a review (Kristrom and Riera 1996) of empirical studies of a variety of environmental indicators (wetlands, forests, water quality, etc ...) it was found that preferences for public environmental quality declined as a proportion of income with increased income levels.

[25] More controversially, the studies also examine the effects of different racial characteristics. In terms of exposure, Brooks and Seith (1997) find that levels are 72% higher for African Americans than whites. However, the econometric results which control the effects of other variables (i.e. percentage urban population) are ambiguous. Brooks and Seith (1997) find that exposure increases with the proportion of African-American residents, while Arora and Cason (1996) find that emissions decrease with the proportion of 'non-white' households until a high proportion is reached.



However, in the econometric analyses undertaken in both studies they do not find that income levels are systematically related to exposure to toxic releases (except for the very highest income groups) when the effects of other factors are included in the analysis. This implies that while there is some degree of (negative) correlation between levels of exposure and income levels, this relationship is largely attributable to other related factors such as the relative income levels of urban versus rural household levels.

Exposure to Pollution from the Transport Sector

Due to the diffuse nature of most environmental damages from residential and industrial energy consumption, it is not surprising that little work has been carried out on the social distribution of its impacts. However, there has been some work on exposure and health impacts from pollution originating from vehicles. For instance, Read (1994) reports on studies carried out in the United States and in Germany which find that children are particularly vulnerable. Whitelegg (1993) reports on a study which found that pre-adolescent children, people over 65, and pregnant women were most at risk. Thus, these groups are likely to benefit most from ETR that shifts people out of cars and onto other modes of transport.

While interesting, such studies do not distinguish between different socio-economic groups, but rather try to identify those members of society who are most likely to be at risk from vehicular pollution emissions and not the extent of exposure *per se*. Given differences in the density of car use it is clear that rural households will tend to benefit rather less than urban households. (For evidence from two rather dated studies in the United States see Harrison 1974 and Gianessi *et al* 1979.) Thus to some extent relative wealth of urban and rural households will capture some of these effects, but there are few studies which have looked systematically at exposure to pollution across socio-economic classifications.

To cast some initial light on the degree of exposure for different socio-economic classifications within individual cities, data was obtained on air quality (ozone, NO_x and

SO_x) at borough level in London (SEIPH 1996). This was compared with average earnings and average house price. (See Table 6.) The former gives a very rough indication of the degree of exposure in the area surrounding the work-place, while the latter gives some indication of the degree of exposure in the area around the home. While the variables are far from ideal,[26] alternatives are not readily available.

Table 6

Economic Wealth and Air Quality in London Boroughs					
	Avg Weekly Earnings	Avg 2BR House	Air Quality (no. of 'very good' days)[27]		
	(FT Employees - £ per Week)	(2nd Q 1996 - £1000's)	Ozone	NOx	SOx
City of London	653	NA	333	208	341
Barking	422	45.5		325	307
Bexley	378	59.4	325	330	330
Brent	387	67.0		120	
Bromley	347	75.7	336		
Ealing	412	90.1	264	261	269
Enfield	375	61.4		203	
Greenwich	367	69.0	312	163	329
Hackney	509	68.7	336	302	
Hillingdon	456	65.5	90	275	89
Islington	477	127.6		30	
Kingston	409	82.1	253		
Southwark	453	73.6	322	217	342
Sutton	384	65.8	189	266	
Tower Hamlets	496	92.1	323	194	322
Wandsworth	392	103.5	340		307
Westminster	510	133.1	254	295	252
Sources: Earnings from UK ONS (1996), House Prices from LRC (1996) and Air Quality from SEIPH (1996).					

It was found that in two out of three cases (ozone and SO_x) air quality - measured in terms of the number of days in which pollution concentrations were in the 'very good' band established by the EU - was positively (but very weakly) correlated with the average weekly earnings and house prices. (See Table 7.) To a limited degree this implies that 'poorer' areas suffer disproportionately from exposure to air pollution and would, thus, benefit disproportionately from improvements in air quality arising from reduced vehicle traffic. Moreover, since poorer households are much less likely to own cars, they contribute very little to the adverse environmental and health impacts from vehicular air pollution. In 1995-1996 only 18.2% of households in the lowest income decile in the UK had a car, while the

[26] The earnings data would, of course, omit the majority of the population who are not in full-time employment and the house price data is only a very imperfect proxy for relative wealth of residents of neighbourhoods. Moreover, comparing both sets of data with air quality indicators at this level of aggregation is clearly inadequate since there can be huge variations in both environmental and socio-economic conditions within individual boroughs. Thus, figures are at most illustrative.

[27] Only background readings are included. Observations from roadside readings are excluded since they are fewer in number and not comparable with background readings. Boroughs with more than one reading were averaged.

figure for the top three deciles were all greater than 90% (UK ONS 1996). Thus, to a great extent poorer households are being exposed to pollution generated by richer households.

Table 7

Correlation Coefficients Between Economic and Environmental Indicators		
	Earnings	House Prices
Air Quality	(FT Employees £ per Week)	(2nd Q 1996 - £1.000s)
Ozone	0.0851	0.1151
NO_x	-0.1288	-0.2057
SO_x	0.0303	0.0103

Environmental Effects from the Agriculture Sector

The distribution of environmental benefits from ETR in the agriculture sector takes two primary forms: improved ambient environmental quality and reduced exposure to chemical residues in agricultural products themselves.[28] With respect to ambient environmental benefits, the two most important consequences are probably in terms of water quality and the effect of the sector on natural landscapes. Other factors such as the contribution of ammonia emissions from livestock to acid rain can be significant in some areas.

The contamination of ground and surface water caused by fertiliser and pesticide application seem to be major hazards to human health.[29] To some extent all members of society benefit from unpolluted clean water. However, those who are near polluted surface waters (or source their water from surface waters polluted by agricultural runoff) would benefit more from reduced discharges from agricultural runoff. In particular, communities in countries that do not have universal water treatment, will enjoy important health benefits from reduced agricultural pollution.

In communities with water treatment facilities the financial benefits may be considerable. For instance Ward *et al* (1993) estimated that substituting one herbicide with another (isoproturon with fenoxaprop-ethyl) saved £800 million in water treatment capital costs and £80 million in annual operating costs. It has also been estimated that removing pesticides from drinking water costs £800 million per year (WWF/CPRE 1996). Since water connection fees and tariffs represent a higher proportion of expenditures for poorer households, environmental taxes may be progressive insofar as water charges are reduced, disproportionately benefiting poorer households. (See Herrington 1997).

Adverse ambient environmental effects from agricultural production will also reduce the value of the countryside environment for non-consumptive uses. For instance, opportunities for recreation may be adversely affected and the contamination of oceans, seas, lakes and fresh water by effluents from agriculture contributes to the destruction of wildlife and their

[28] Health effects arising from the effects of ETR on the price (and thus consumption levels) of food, rather than its quality, will be discussed below.

[29] The UK Department of the Environment (1986) estimated that 1 million people in the UK were regularly receiving drinking water that exceeded the 50 mg of nitrates per litre target set in an EC directive. In 1990 192 water courses in the UK exceeded the limit (WWF/CPRE 1995).

habitats (Miranowski 1983, Liapis 1994, Libby and Boggess 1990). The distributional consequences are complex. On the one hand, rural residents are likely to benefit more than urban households from the enhanced environment which will accompany effective ETR. On the other hand, amongst urban residents, the major beneficiaries of these opportunities may well be rich households due to their greater ability to gain access to improved rural landscapes. However, WWF/CPRE (1996) report that approximately 60% of the population "visit" rural landscapes at least once a year, indicating that the benefits of improved landscapes are relatively widely distributed.

Similarly, both rural and urban residents are likely to benefit from reductions in chemical residues in foods. However, the benefits from reduced exposure to pesticides (in application) will be exclusively enjoyed by rural households. Therefore, it is likely that reduced environmental pollution will benefit rural residents more than their urban counterparts. Furthermore, since people employed directly in the agriculture sector are likely to be poor relative to the average, the overall distributional effects of controlling agricultural pollution will be progressive. For instance in the UK in 1996 male "manual" workers in the agricultural sector had an annual income that was 81% of the national average for all workers (ONS *NES* 1996). The figures for male "non-manual" (78%) and female "manual" (93%) workers were also lower than the average. (The sample size was too small to give a reliable figure for female "non-manual" workers in the sector.) Thus, the distribution of health benefits in the sector may be progressive for society as a whole, particularly since occupational health concerns can be significant. For instance, mortality rates from exposure to pesticides vary from 1% to 9% of cases presented for treatment. The global number of such cases each year has been estimated at between 1 to 3 million (WHO 1992).

IV.B Distribution of Social Effects of ETR

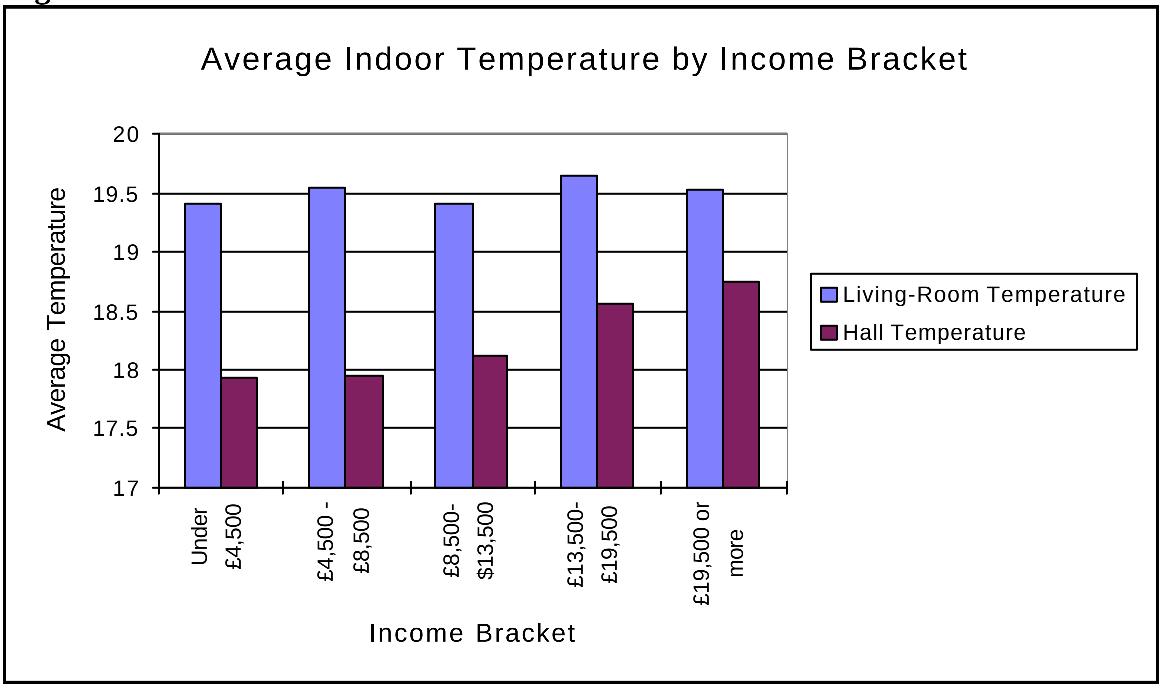
The social effects of environmental tax reform are manifold, and in this section some of the main issues - such as access to dwelling warmth, avoidance of traffic accidents, and access to a sufficiently nutritious diet - will be reviewed. In addition, the employment effects which are likely to arise from ETR will be reviewed.

Energy

As noted in Section III, the fiscal consequences of taxes on domestic fuels - whether in the form of sales taxes or through a broader carbon or energy tax - are generally regressive, unless accompanied by a form of revenue recycling which explicitly favours the poorest. Moreover, there are good reasons to believe that the consequences in terms of health are also regressive. For instance, in their analysis of the effects of VAT on domestic fuel in the UK Johnson, McKay and Smith (1990) estimate that the lowest deciles will reduce fuel consumption by 9.6% and 9.5%, while the average decrease for all households is only 4.1%. In a later study using the same model, Crawford, Webb and Smith (1993) found that following the introduction of VAT on domestic fuel at 17.5% the lowest quintile would reduce energy consumption by just over 9%, while the richest quintile would reduce consumption by only 1.0%.

The implications of this for the poorest households are quite significant since average indoor temperatures in the lowest socio-economic categories of household are already lower than in richer households. Figure 8 presents data on average indoor temperature levels for different income brackets in the UK in 1991 (UK DOE 1996). While living-room temperatures do not vary markedly, hall temperatures (which are a good proxy for average house temperatures) exhibit a marked increase with income levels. Moreover, the variance is much higher for the lowest-income bracket, meaning that some households live in extremely cold dwellings. An earlier report (UK DOE 1991) found that well over one in five households in the lowest income bracket had an average indoor temperature of less than 12^0. In many cases residents of these households are least able to withstand such temperatures. For instance, the SAP rating for private tenants over the age of 85 is 12.2^0, one-third of the average for all households (UK DOE 1996).

Figure 8



Moreover, as noted in Section III lower-income households are particularly ill-equipped to undertake the kinds of investment needed to increase energy efficiency (i.e. invest in more efficient heating equipment and insulation measures), and as such the drop in energy consumption is more likely to be directly reflected in corresponding drops in temperature. For such households any drop in indoor temperature will be reflected not only in terms of reduced comfort but, in many cases, also in terms of adverse health impacts. As such, the drop in temperature should not be seen as a reflection of household consumer preferences in the face of changing prices, but rather as a reflection of their inability to meet basic needs. Many poorer households spend as much on fuel as they are able to and any increase in price can only be reflected in reduced consumption.

Transport

Similar issues arise in the case of transport, where the principal non-pollution benefits from ETR are related to reduced noise, congestion, and accidents from the use of private road vehicles. It has been estimated that in the UK the marginal external costs can be as high as £19.1 billion for congestion, £2.6-£3.1 billion for noise, and £2.9-£9.4 for accidents (Maddison *et al* 1996). Unfortunately, there is little evidence on the distribution of these benefits by different socio-economic classifications.

However, in the case of congestion the primary beneficiaries are likely to be the users of the roads themselves. Since some of the costs of congestion are internal, some of the benefits from its reduction will also be internal. For instance, an American study found that internal 'user time' costs per vehicle mile in the United States were more than five times greater than external congestion costs (Litman 1995). Moreover, since car drivers and passengers are generally wealthier than average this implies that the benefits are likely to be regressive. However, users of public transport modes - who are generally less wealthy than car drivers[30] - will also benefit to some extent, through reduced journey times.

Table 8

Fatalities from Motor Vehicle Traffic Accidents in the UK in 1994		
Vehicle-Vehicle Accident		
Driver of Vehicle	578	17.88%
Passenger of Vehicle	268	8.29%
Motorcyclists	227	7.02%
Vehicle-Cycle Accident		
Cyclist	136	4.21%
Other	3	0.09%
Vehicle-Pedestrian Accident		
Pedestrian	994	30.75%
Other	2	0.06%
Other Accidents		
Driver of Vehicle	411	12.72%
Passenger of Vehicle	261	8.08%
Motorcyclist	142	4.39%
Other	210	6.50%

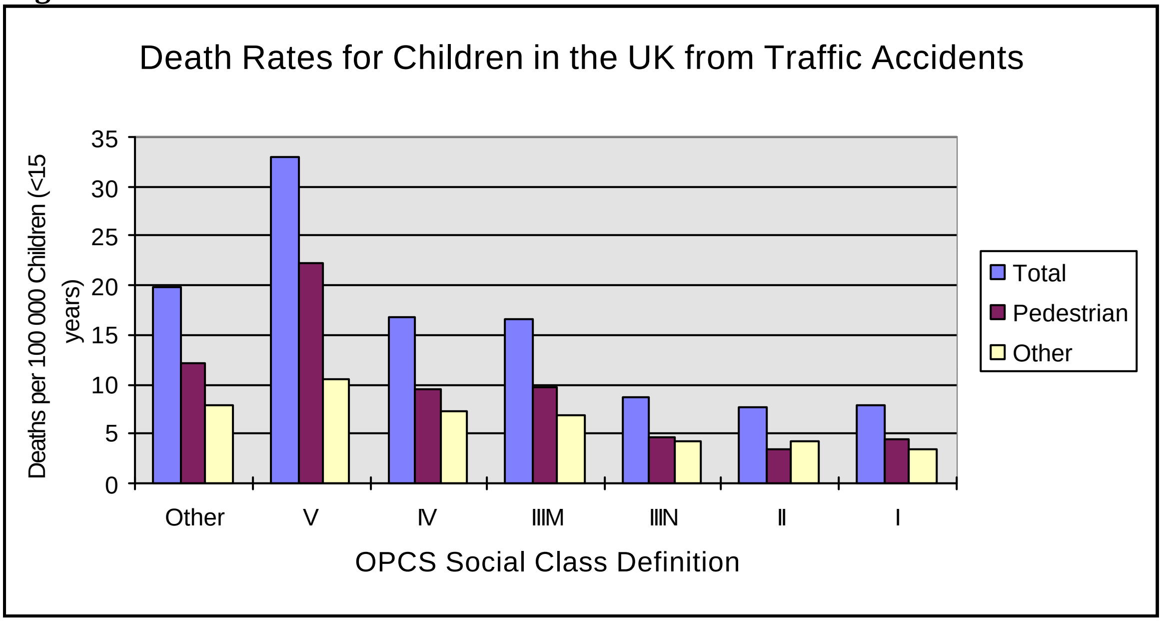
The same is true of accidents to the extent that a large proportion of accidents are between pairs of vehicles, and thus the distribution of benefits would be largely proportional to the costs borne. However, users of public transport, cyclists and pedestrians will also benefit to a great extent since they currently bear some of the costs. For instance, in the UK in 1994 136 (4.2%) and 994 (30.8%) of the 3,232 traffic accident fatalities involving motor vehicles were cyclists and pedestrians respectively (UK OPCS 1996). (See Table 8.)

Accident rates by socio-economic classification are more difficult to quantify. In a recent study by Roberts and Power (1996), death rates for children from traffic-related accidents

[30] The percentage of expenditures on public transport fares relative to expenditures on private motoring costs was just under 40% for the lowest decile in the UK in 1995-1996, while the figure for the highest decile was 16% (UK ONS 1997).

are compared by OPCS-defined occupational social classes for the period 1989-1992. (See Figure 9.) The figures are lower for the more advantaged classes (i.e. classes I and II) relative to the more disadvantaged classes (i.e. classes IV and V).[31] (The "other" category comprises households where the household head is "unoccupied", in the armed forces, or where insufficient information was provided.) Children in class V are over four times as likely to die from a traffic-related accident than children in classes I or II. Moreover, the degree of inequality in vulnerability appears to be increasing. For instance, the ratio of death rates for classes V and I was 1.8:1 in 1979-1983, but 4.2:1 in 1989-1992. For pedestrian deaths the ratio was 4.3:1 in 1979-1983 and 5.1:1 in 1989-1992.

Figure 9



Some of the most important social effects of ETR in the transport sector are likely to arise from changes in access to mobility. To the extent that environmental taxes on private motor transport are indicative of a more general shift away from land use patterns based around the car relative to other transport modes, this would tend to benefit poorer households who do not own cars and whose mobility has been restricted by the emphasis on the car in recent decades. To the extent that 'mobility' is a wider social need, the social consequences of such development patterns is significant. (See ECMT/OECD 1994 and Whitelegg 1993.). This argument is particularly relevant in countries such as the UK with the deregulation of public transport services and the increasingly extended nature of development patterns.

However, while it can be argued that some groups may find their mobility enhanced following the introduction of ETR, there is little question that rural inhabitants will be adversely affected. Car ownership in some rural areas is effectively forced, and thus rural households have few alternatives but to bear the costs (ECMT/OECD 1994). For instance, Casler and Rafiqui (1993) find that rural households would be hardest hit by petrol taxes in Australia, spending almost 7% of total expenditures on petrol (direct and indirect

[31] The classifications are as follows: I professional occupations, II managerial and technical occupations, III skilled (N) non-manual and (M) manual occupations, IV partly-skilled occupations, and V unskilled occupations.

consumption), relative to just over 5% for urban households. Johnson, McKay and Smith (1990) estimate that following the introduction of an increase in the petrol duty in the UK, rural households would face the largest increase in taxes *and* the largest decrease in petrol consumption.

Agriculture

The distribution of the social consequences of ETR in the agriculture sector may also be severe. Indeed, it has been estimated that the poor in the UK would have to spend more than 40% of their incomes on food in order to purchase a 'healthy diet' as set out in government guidelines (NCC 1995). This suggests that the food that poor households currently purchase with 25.2% of their income (see Section III) does not provide adequate nutrition.

Thus, without any improvements in their economic means following ETR in the agriculture sector, these groups would simply have to absorb the price increases, reduce calorie intake or switch to an even less nutritional diet. Johnson, McKay and Smith (1990) estimate that a 5% increase in food prices would result in a decrease in food consumption of approximately 4% for the lowest decile. While other deciles face similar decreases, the consequences in nutritional terms are likely to be much greater for lower-income households since their diets are already insufficient. Perhaps most distressingly, households with children and retired persons - those most vulnerable to dietary deficiencies - face the biggest falls in consumption.

The situation may be worse still, since poor households appear to be dependent on cheap foods that are high in calorific content but may be of limited nutritional value (NCC 1995). Comparing relative expenditures on different food types (UK ONS 1996) with their relative calorific contents for a given cost (NCC 1995) reveals that there is some bias amongst low-income households towards foods which are relatively inexpensive but filling (i.e. potatoes and bread), relative to others which are nutritious but more expensive per calorie (i.e. fruit and vegetables). (Table 9.) To some extent, it would appear that low-income households are already being forced to substitute foods of relatively lower nutritional value in an effort to meet calorific requirements.

Table 9

The Calorific Content of Selected Foods and Expenditure Patterns by Decile

	Pence/ 100 Cals	As % of Food Expenditures		As % of Total Expenditures	
		D1	D10	D1	D10
Potatoes	3p	3.95%	2.98%	0.83%	0.27%
Bread	4p - 7p	5.79%	4.31%	1.22%	0.39%
Fruit	19p - 30p	5.96%	7.31%	1.25%	0.66%
Vegetables	20p - 103 p	7.85%	8.91%	1.65%	0.81%

Whether or not ETR has a significantly adverse effect on the diets of the poorest households will depend upon the correlation between the environment-intensity of different food types, the proportional consumption of different food types, and their nutritional value. In some cases, efforts to address environmental problems may also yield health benefits as healthier food types become relatively less expensive. Similarly, within individual food types ETR may result in cultivation practices with positive health consequences (e.g. fewer chemical residues). However, this is clearly not always the case. For instance, the primary motivation behind the application of some inputs with adverse consequences for the ambient environment has been concern about consumer health.

Employment Costs

Other important social effects arise from the sectoral and regional dislocation which accompanies any significant policy reform. While in some cases these effects may be temporary, as capital and labour shift from particular sectors and areas, the transitional costs for certain groups can be considerable in the interim (OECD 1994b). In particular, the employment effects of changes in the economy accompanying ETR may be considerable.[32]

These costs will be particularly high if some regions specialise in a narrow range of environment-intensive sectors, since other employment opportunities will be few. Moreover, although the costs are likely to be highest in the short- and medium-term, they may persist as long-term unemployment becomes entrenched in particular regions. Conversely, employees in other areas are likely to benefit, finding demand for their skills, resources and products very high. DeWitt *et al* (1991 in OECD 1994b) find that the regional effects of a carbon tax can vary by as much as 50% in the nine census regions in the United States. Osten *et al* (1991 in OECD 1994b) find similar results for Canada, with Alberta's economy hardest hit due to its concentration in carbon-intensive sectors.

Using a model of the UK economy, Barker (1997) looks at the sectoral and employment effects of a carbon/energy tax in the EU and finds quite different effects. Most pertinently, while a number of sectors show falls in aggregate output (i.e. the energy sectors, car manufacturing, pharmaceuticals and chemicals, transport, textiles and apparel, etc..), none of these sectors show falls in employment. This is largely a consequence of the assumption that the revenue is recycled via reduced lower non-wage labour costs. Thus, even in those sectors in which output falls, incentives to increase employment are more than sufficient to counterbalance this effect. Similarly, in a comparison of different policy scenarios (and model specifications) Mabey and Nixon (1997) show that the employment effects of introducing a carbon tax sufficient to stabilise emissions at 1990 levels differ significantly depending on whether the revenue is recycled via reduced income taxes or employers' national insurance contributions. In the latter case employment rises by as much as 3% for one of the applications.[33] These results highlight the importance of using appropriate recycling to mitigate any adverse social consequences of ETR. This is particularly important in the EU where labour is much less mobile than in North America.

These issues are of even greater importance in agriculture since the usual assumption of mobility is less applicable than it is in many other sectors in the economy for a number of

[32] For discussions of the effects on other groups (i.e. shareholders, etc ...) see Smith (1992) and OECD (1994b).

[33] Although discrepancies in the results depending upon the model specification reveal the inherent uncertainty associated with estimating such effects.

reasons: the skills required in the sector are less readily transferable; workers in the sector are often quite far removed from other job opportunities; the average level of education in the agricultural labour force is less than that in other sectors; and, governments of industrialised countries have traditionally invested less in rural education and in training people in rural communities for alternative employment opportunities. As a result, people employed on farms have fewer alternative employment opportunities and the effects of environmental taxation on agricultural employment and income are likely to be much higher.

Thus, Wu *et al* (1995) predict that the imposition of nitrogen or irrigation water taxes set high enough[34] to reduce nitrate losses by 5% cent would cause farm income losses of between 20 and 35%. Liapis (1994) reports that a 50% tax on fertiliser inputs would reduce EU farm incomes by 7.3%, which is lower down on the spectrum of estimated farm income losses. Rendleman (1991) predicts a reduction in returns to labour of about 0.7% from a nitrogen tax of 150% in Norway. However, the small size of this reduction in the returns to labour is mainly due to the assumption of high labour mobility.[35]

Employees in downstream manufacturing sectors will also be affected. McCorriston and Sheldon (1989) noted that while the burden of a fertiliser tax in the UK falls largely on the farmers, fertiliser firms are also affected. Rendleman (1991) evaluates the wider economic effects of policies such as input taxes that would bring an across-the-board 75% cent reduction in agricultural chemical use for the US. He finds that the reduction would cause significant production changes especially in the farm sectors and downstream industries. Feed grains and oilseeds sectors suffer the highest (-20.4%) contraction in output. Other farm sectors also suffer significant contractions. However, manufacturing output also falls by -0.8%. In particular, the livestock and feed grains/oilseeds processing industries would suffer output contractions by -2.4% and -7.9% respectively. Conversely, Tobey and Reinert (1991) show that an agricultural policy reform which causes a reduction in fertiliser use will increase output and employment in the manufacturing sector.

However, while it is true that ETR in the agriculture sector will be felt in terms of reduced employment in some upstream and downstream industries, its effects on the means by which agricultural goods are produced may result in increased employment in parts of the sector itself since in many cases polluting inputs are substitutes for labour inputs. For instance WWF/CPRE (1996) report that “integrated crop management” is labour-intensive. Lampkin (1990) provides data for organic farming in Switzerland, Germany and the Netherlands indicating that it is generally more labour-intensive than conventional farming, but it is not clear that the effects of different crop mixes are adequately addressed in the studies cited. Thus, while the precise consequences will depend upon the reform introduced, it is quite likely that by targeting some of the more environment-intensive crops and production processes, more labour-intensive systems (such as organic agriculture and

[34]It is estimated that for 5% reduction in nitrate loss the tax rates 85% and 313% have to be imposed respectively in the North and South subregions.

[35] Dubgaard (1990) predicts that agricultural land rent in Norway would decrease by 25% or more with the use of nitrogen taxes of 150%. The study by Rendleman (1991) cited above also predicted a decrease in land rent of 34% and 21% for land under fruit and vegetable production and for land under feed grain production, respectively.

integrated pest management) will benefit. This may mitigate some of the adverse social consequences discussed above.

IV.C Conclusions

The distribution of the effects of environmental taxes depend largely upon the precise reform introduced as well as the environmental indicator. For global pollutants such as carbon dioxide and other greenhouse gases from all three sectors the issue is only relevant to a limited extent at the national level. However, for other concerns (i.e. exposure to local airborne pollution from the transport sector, polluted agricultural runoff in surface waters, improved environmental amenities) poorer households stand to gain a great deal from the environmental benefits arising from many forms of ETR. Whether or not their gains exceed those realised by richer households varies upon the environmental indicator and the policy reform, but contrary to common assertion it is clear that environmental benefits are not a 'luxury' demand enjoyed only by the rich.

In social terms, there is little question that some lower-income households may face significant hardship for many forms of ETR. For instance, dwelling temperatures may fall for members of households which are already suffering from the cold. Similarly, diets may suffer for those who already have an insufficiently nutritious diet. ETR in the transport sector may result in some poorer households in outlying areas having their mobility restricted. Conversely, other benefits such as reduced exposure to traffic accidents and traffic noise are likely to be realised disproportionately by poorer households.

V. Reconciling the Economic, Environmental and Social Effects of Environmental Tax Reform

The preceding sections illustrate the importance of addressing distributional consequences when introducing environmental tax reforms. Both the costs and benefits from introducing ETR differ widely across different groups within society. Moreover, as ETR starts to play a larger role in the economy these differences will become more marked. Thus, it is important to ensure that ETR is not only economically and environmentally efficient, but also fair. If this is not the case then proposed measures will neither be politically feasible nor socially desirable.

Unfortunately, in many cases it appears that the twin objectives of fairness and efficiency are often in conflict, with the most efficient measures often being the most inequitable, and the most equitable measures often being the most inefficient. For instance, in the study cited above by Symons, Proops and Gay (1994), due to the environmental inefficiency of the non-regressive tax reform, the carbon tax rate would have to be £444/ton. This is considerably higher than would be the case if social objectives were ignored entirely (£277/ton), while still meeting the revenue neutrality constraint. Similarly, Johnson, McKay and Smith (1990) show that the most equitable form of redistribution of energy taxes (i.e. lump-sum payments) is the most inefficient. The problem is that some of the measures which have been proposed (and applied) in order to mitigate the adverse distributional effects of

ETR (i.e. transfer payments) often undermine the environmental effectiveness of the measure itself.

However, this apparent contradiction between the equity and efficiency aspects of ETR can be overcome. The combined effects of the existence of market failures and barriers in the market for many goods which are environment-intensive and which are of considerable importance to poorer households, as well as the large amount of revenue raised by many environmental taxes, raises the possibility of introducing specific measures which increase economic efficiency, address environmental degradation, and reduce distributional inequality. ETR which is both effective and fair not only needs to change prices in the market, but also to change the way households respond to changing prices in the market.

For instance, it is clear that there are significant efficiency gains to be realised in the domestic energy sector. Nonetheless, such opportunities may not be realised in the existing market for energy and relative price changes may not be sufficient to encourage more efficient energy use, both in terms of overall fuel use and in terms of the fuel mix. This is particularly true for poorer households. Thus, the exclusive use of price mechanisms may not be appropriate. The key to a successful policy in terms of both efficiency and equity objectives rests with the ability of policy makers to institute a programme which changes the characteristics of the market itself.

Barker and Johnstone (1993) analysed the effects of the joint introduction in the UK of a carbon/energy tax and a government-sponsored home insulation programme which encourages energy conservation amongst low-income households. This was compared with a policy which uses standard lump-sum benefits to redress any adverse welfare effects on low-income households. The size of compensation was that amount of support required to return low-income households to their pre-tax consumption patterns. Assuming that low-income households respond to changed relative prices in the same way as households on average, they would require compensation equal to £282.1 million in the year 2000 (Table 10, Row 1).

Table 10

The Distributional Effects of a Carbon Tax Under Different Recycling Scenarios

	1995	2000	2005
Price Response Only	141.6	282.1	170.4
Plus Insulation Programme	48.4	-341.4	-725.2

Source: Barker and Johnstone (1993). £ mn (1992 prices) difference in expenditures from base case for low-income households.

Given the relatively greater importance of capital constraints and tenancy disincentives amongst lower income households, this figure certainly underestimates the extent of compensation. If, however, an energy-efficiency programme were instituted rather than direct transfer payments for fuel consumption, the level of compensation required would fall. A simulation of such a programme was introduced into the model in order to evaluate its relative merits.[36] Since the effect of the programme would be to encourage conservation, low-income households could conceivably consume less fuel and not require any compensation in order to retain welfare levels. In fact, they would experience a net increase in disposable income of £341.4 million by 2000 and £725.2 million by 2005 (constant 1992 £).

The difference between the effects of the two programmes results from the fact that under the lump-sum scheme the government will pay (indefinitely) for continued inefficient energy consumption patterns of low-income households, whereas under the energy-efficiency programme the government is investing in measures which reduce expenditures (for the working-life of the investments). The overall increase in efficiency is illustrated by the fact that the government's own payback period for the energy-efficiency programme relative to lump-sum benefit redistribution is just under four years. Thus, under well-designed environmental tax reforms, efficiency and equity can be complementary objectives, even for those sectors in which the financial incidence of taxes is likely to be regressive.

Balancing efficiency and equity objectives is also possible when dealing with environmental problems in the agriculture sector. For instance, it has been estimated that the market distortions created by the Common Agricultural Policy (CAP) have increased food prices for the average UK household by as much as £10/week (SSF/CPRE 1995). To the extent that distortions arising from the CAP have also resulted in environment-intensive cultivation practices, both efficiency and equity objectives can be achieved through the removal of subsidies with adverse environmental consequences. Moreover, within the agriculture sector itself, it has been argued that the CAP has benefited larger farms with more intensive cultivation practices disproportionately relative to smaller farms, a point recognised by the Commission itself prior to the 1992 MacSharry "reforms". (See Swinbank and Tanner 1992 and Whitby 1994.)

However, there is little question that the distributional and social effects of introducing other measures which increase food prices and reduce agricultural employment may be significant, and thus the mechanisms designed to alleviate the adverse effects should form an important component of their design and implementation. At the most basic level, social security benefits must be fully indexed to the actual cost of food for low-income households and constitute a pre-condition for the formulation of policies to control agricultural pollution. Additional transfer mechanisms are also required to compensate vulnerable households which are not eligible for social security benefits but which nonetheless would be adversely affected by higher food costs. For instance, given the vulnerability of children to dietary deficiencies Johnson, McKay and Smith (1990) advocate increased access to free school meals.

Such compensatory policies may not only be equitable, but efficient since they are in both the short- and long-term national interest. For instance, a NCC report states that nutritional inadequacies exist in the diets of many families living on state benefits in the UK (National

[36] According to a source in the electricity industry, a £1.1 billion once-and-for-all programme of conservation measures geared toward low-income households instituted in 2000 would result in a reduction of 170 million therms in low-income quintile energy use in 2001 and all subsequent years.

Consumer Council (NCC) 1995). A detailed study conducted in Leeds shows that 85% of the variance by ward in morbidity before age 65 can be explained in terms of material deprivation (NCC 1995). The long-run efficiency of ensuring that vulnerable households are compensated for the adverse consequences which may accompany higher food prices is reflected in the fact that there appears to be a high correlation between low birth-weight and the nutrient intake of expectant mothers. Moreover, the adverse health effects of low birth weight may persist for years (NCC 1995). Thus, any financial savings which accrue to the Department of Social Security due to inadequate compensation is likely to be more than counterbalanced by higher expenditures borne by the National Health Service, to say nothing of the costs borne by low-income households in terms of quality of life.

Given the present level of subsidies received by farmers in the EU, some contraction in output and some dislocation amongst agricultural workers is inevitable. However, measures should be introduced which maximise the environmental benefits and minimise the social costs arising from such structural changes. In particular, it is important to recognise that there may be huge opportunity costs associated with improvement in the environment without simultaneously correcting other sources of market failure. For instance, some of the studies cited above (for example Weaver *et al* 1996 and Taylor *et al* 1992) indicate that significant reductions in agricultural pollution could be achieved through changes in cultivation practices at little or no cost through the removal of perverse subsidies. Schemes could be used to transfer income back to farmers while at the same time encouraging more environmentally friendly practices.

Conversely, the distributional effects of ETR in the transport sector are likely to be less adverse than in the residential and agricultural sectors. However, even in this case the efficiency of ETR can be improved by more enlightened forms of recycling since, as in the other two cases, there are numerous market failures and barriers in the sector. In particular, households will respond more elastically to increased petrol prices if their public transport options are improved. Not only will this allow for greater reductions in emissions from the sector, but the social benefits may be important in terms of increased mobility for particular groups of households who rely upon public transport and who have suffered in recent decades from patterns of development which revolve around the use of private automobiles.

VI. Conclusion

This report has examined the distributional effects of environmental tax reform, focusing on measures designed to reduce pollution emissions from the energy, transport and agriculture sectors. The report emphasises that in at least two cases (energy and agriculture) ETR is likely to have adverse distributional consequences, at least in terms of direct tax burdens. This arises from the fact that domestic energy consumption and food are basic needs, consumed in relatively greater proportion by lower-income households. Moreover, the distributional consequences in these sectors may be even more adverse when it is recognised that poorer households may be less able to adjust consumption patterns in the face of changing prices. However, it has also been emphasised that since environmental taxes (unlike other environmental policy instruments) raise significant revenue, some of the adverse distributional consequences of ETR can be overcome by lowering other taxes or increasing expenditures.

The report has also examined the distribution of the environmental and social consequences of ETR in the three sectors. While the effects depend largely upon the precise reform introduced, it is clear that in many senses (i.e. exposure to pollution, improved environmental amenities) poorer households stand to gain a great deal from the environmental benefits arising from ETR. Whether or not their gains exceed those realised by richer households varies upon the environmental indicator and the policy reform, but it is clear that many environmental benefits are not enjoyed disproportionately by wealthier households. In social terms, there is no question that some lower-income households may face significant hardship. For instance, dwelling temperatures may fall and dietary requirements may not be met satisfactorily. Conversely, other reduced benefits such as exposure to traffic accidents are likely to be realised disproportionately by poorer households.

However, the distributional consequences which arise from some environmental tax measures should not be used as a pretext to delay or restrict their introduction. Instead, measures to mitigate their adverse distributional consequences should be integrated fully into the policy reform. Thus, rather than addressing distributional consequences in an *ad hoc* and *ex post* manner[37] they should be addressed systematically, recognising the characteristics of the sectors and households affected. Given the prevalence of market imperfections in many environment-intensive sectors and the particular constraints faced by low-income households, in many instances it will be possible to realise both environmental and distributional objectives in a manner which is economically efficient.

[37] The recent Labour initiative to provide as much as £50 for some low-income households to cover winter fuel costs is indicative of such a policy, particularly since the concomitant fall in VAT in insulation only applies to a small sub-set of households which are involved in government energy efficiency schemes.

VII. References

Arora, Seema and Timothy Cason. 1996. 'Do Community Characteristics Determine Environmental Outcomes?' Paper Presented at the EARE Conference, Lisbon, September.

Barker, Terry and N. Johnstone. 1993. 'Equity and Efficiency in Policies to Reduce Carbon Emission in the Domestic Sector' in *Energy and Environment* Vol. 4, No. 4.

Barker, Terry and N. Johnstone. 1998. 'Competitiveness and the Carbon Tax' in Terry Barker (ed.) *Environmental Policies and International Competitiveness*. Brookfield Vermont: Edward Elgar.

Barker, Terry. 1997. 'Taxing Pollution Instead of Jobs' in T. O'Riordan (ed.) *Ecotaxation*. London: Earthscan.

Barker, Terry., S. Baylis and P. Madsen. 1993. 'A UK Carbon/Energy Tax: The Macroeconomic Effects' in *Energy Policy*, Special Issue, April.

Baumol, W. and W.E. Oates. 1988. *The Theory of Environmental Policy*. Cambridge: CUP.

Boadway, R. W. and N. Bruce. 1984. *Welfare Economics*. Oxford: Basil Blackwell.

Bohm, Peter. 1997. 'Environmental Taxation and the Double Dividend' in T. O'Riordan (ed.) *Ecotaxation*. London: Earthscan.

Brechling, V., D. Helm and S. Smith. 1991. 'Domestic Energy Conservation: Environmental Objectives and Market Failures', in D. Helm (ed.) *Economic Policy Towards The Environment*. Oxford: Blackwell.

Brechling, Vanessa and Stephen Smith, 1994. 'Household Energy Efficiency in the UK' in *Fiscal Studies* Vol 15, No. 2.

Brooks, Nancy and Rajiv Sethi. 1997. 'The Distribution of Pollution: Community Characteristics and Exposure to Air Toxics' in *Journal of Environmental Economics and Management* Vol 32, pp. 233-250.

Casler, Stephen D et al. 1993. 'Evaluating Fuel Tax Equity: Direct and Indirect Distributional Effects' in *National Tax Journal* Vol 46, No. 2.

Cornwell, Antonia and John Creedy. 1997. *Environmental Taxes and Economic Welfare: Reducing Carbon Dioxide Emissions*. London: Edward Elgar.

Crawford, Ian, Stephen Smith and Stephen Webb. 1993. 'VAT on Domestic Energy' London: IFS Commentary No. 39.

DeWitt, Diane E. et al. 1991. 'Who Bears the Burden of Energy Taxes?' Discussion Paper No. QE91-2. Washington DC: Resources for the Future, March.

Energy Efficiency Office (1990) *Energy Use and Energy Efficiency in the UK Domestic Sector*, HMSO, London.

European Conference of Ministers of Transport/OECD. 1994. 'Internalising the Social Costs of Transport'. Paris: ECMT/OECD.

European Environment Agency. 1996. 'Environmental Taxes: Implementation and Environmental Effectiveness'. Copenhagen: EEA.

Ferguson, Malcolm and David Taylor. 1996. 'Greening Vehicle Excise Duty'. London: Institute for European Environmental Policy for RSPB.

Gale, Robert et al. 1995. *Green Budget Reform*. London: Earthscan.

Gee, David. 1997. 'Economic Tax Reform in Europe' in T. O'Riordan (ed.) *Ecotaxation*. London: Earthscan.

Gianessi, Leonard P. et al. 1979. ‘The Distributional Effects of Uniform Air Pollution Policy in the US’ in *Quarterly Journal of Economics* Vol. 93, No. 2, 281-301.
Goodman, A. Paul Johnson and S Webb. 1997. *Inequality in the UK*. Oxford: OUP.
Goulder, Lawrence H. 1994. ‘Environmental Taxation and the Double Dividend: A Reader’s Guide’ NBER Working Paper No. 4896. Cambridge, Mass.
Harrison, David. 1974. *Who Pays for Clean Air?*. Cambridge, Mass.: Ballinger.
Herrington, Paul.1997. ‘Pricing Water Properly’ Europe’ in T. O’Riordan (ed.) *Ecotaxation*. London: Earthscan.
Huang, K.S. 1996. ‘Nutrient Elasticities in a Complete Food Demand System’ *American Journal of Agricultural Economics* Vol 78, 21-29.
Johnson, Paul, Steve McKay and Stephen Smith. 1990. ‘The Distributional Consequences of Environmental Taxes’. Institute for Fiscal Studies Commentary 23. London, July.
Jorgenson, D.W., D.T. Slesnick, and P.J Wilcoxen. 1992. ‘Reducing US Carbon Emissions’ in *Brookings Papers: Microeconomics* pp. 393-454. Washington.
Kneese, Alan and Charles Schultze. *Pollution, Prices and Public Policy*. Washington, D.C: Brookings Institute.
Kristrom, Bengt and Pere Riera. 1996. ‘Is the Income Elasticity of Environmental Improvements Less than One?’ in *Environmental and Resource Economics* Vol 7, No. 1, pp. 45-55.
Lampkin, N. 1990. *Organic Farming*. Ipswich: Farming Press.
Liapis, Peter S. 1994. ‘Environmental and Economic Implications of Alternative EC Policies’ in *Journal of Agricultural and Applied Economics* Vol. 26**,** 241-251.
Libby, Lawrence W and William G. Boggess. 1990. ‘Where Are We and Why?’ in John B. Braden and Stephen B. Lovejoy (eds). *Agriculture and Water Quality: International Perspectives*. London: Lynne Rienner.
London Research Centre. 1996. ‘London House Prices’ LRC Quarterly Bulletin. London: August.
Mabey, Nick and James Nixon. 1997. ‘Are Environmental Taxes a Free Lunch?: Issues in Modelling the Macroeconomic Effects of Carbon Taxes’ in *Energy Economics*, Vol. 19, No. 1, 29-56.
Maddison, D. et al. 1996. *The True Costs of Road Transport*. London: Earthscan.
McCorriston, S. and I. Sheldon 1989. ‘The Welfare Implications of Nitrogen Limitation Policies’ in *Journal of Agricultural Economics*, 40: 143-51.
Miranowski, J.A. 1983. ‘Agricultural Impacts of Environmental Quality.’ In T. L Napier et al (eds) *Water Resources Research: Problems and Potentials for Agriculture and Rural Communities*. Ankeny: Soil Conservation Society of America.
National Consumer Council. 1995. ‘Budgeting for Food and Benefits: Budget Studies and their Application in Europe.’ A Report Published by the National Consumer Council. London.
O’Riordan, T. 1997. *Ecotaxation*. London: Earthscan.
Organisation for Economic Cooperation and Development. 1993. *Agricultural and Environmental Policy Integration: Recent Progress and New Directions*. Paris: OECD.
Organisation for Economic Cooperation and Development. 1994a. *Managing the Environment: The Role of Economic Instruments*. Paris: OECD.

Organisation for Economic Cooperation and Development. 1994b. *The Distributive Effects of Economic Instruments for Environmental* Policy. Paris: OECD.
Organisation for Economic Cooperation and Development. 1996. *Environmental Taxes in OECD Countries*. Paris: OECD.
Palmquist, Raymond B. et al. "Hog Operations, Environmental Effects and Residential Property Values" in *Land Economics*, Vol. 73, No. 1, pp. 114-124.
Parry, W.H. 1995. 'Pollution Taxes and Revenue Recycling' in *Journal of Environmental Economics and Management* Vol. 29, S64-S77.
Pearson, Mark and Stephen Smith. 1991. *The European Carbon Tax: An Assessment of the EC's Proposal*. London: IFS.
Poterba, James M. 1993. 'Global Warming Policy: A Public Finance Perspective' in *Journal of EconomicPerspectives* Vol. 7, No. 4, 47-63.
Poterba, James. 1991. 'Tax Policy to Combat Global Warming' in R. Dornbusch and J.M. Poterba (eds.) *Global Warming: Economic Policy Perspectives*. Cambridge: MIT Press.
Read, Cathy. 1994. 'How Vehicle Pollution Affects our Health' Symposium on Vehicle Pollution and Health. Ashden Trust. London.
Rendleman, C. Matthew. 1991. 'Agrichemical Reduction Policy: Its Effects on Income and Income Distribution' in *The Journal of Agricultural Economics Research* Vol. 43, No. 4, 3-9.
Repetto, R. et al. 1992. *Green Fees: How a Tax Shift Can Work for the Environment and the Economy.* Washington DC: World Resources Institute.
Roberts, Ian and Chris Power. 1996. "Does the Decline in Child Injury Mortality Vary by Social Class?" in *British Medical Journal* Vol. 313, 28 September, 784-786.
Robison, H. David. 1985. 'Who Pays for Industrial Pollution Abatement?' in *The Review of Economics and Statistics* Vol. 67, No. 4, 702-706.
Smith, Stephen. 1992. 'The Distributional Consequences of Taxes on Energy and the Carbon Content of Fuels' in *European Economy: The Climate Challenge - Economic Aspects of the Community's Strategy for Limiting CO2 Emissions* in *European Economy*, No. 51, Brussels.
South East Institute of Public Health. 1996. *Air Quality in London: The Third Report of the London Air Quality Network*. London: SEIPH.
Statistics Canada. 1992. *Family Expenditure In Canada 1992*. Ottawa: StatsCan.
Swinbank, Alan and Carolyn Tanner. 1996. *Farm Policy and Trade Conflict: The Uruguay Round and the CAP Reform*. Ann Arbor. University of Michigan Press.
Symons, Elisabeth, John Proops and Philip Gay. 1992. 'Carbon Taxes, Consumer Demand and Carbon Dioxide Emissions: A Simulation Analysis for the UK' London' in *Fiscal Studies* Vol. 15, No. 2, 19-43.
Taylor, Michael L, Richard M. Adams, and Stanley F. Miller. 1992. 'Farm-Level Response to Agricultural Effluent Control Strategies: The Case of the Willianette Valley.' *Journal of Agricultural and Resource Economics* Vol. 17, No. 1, 173-185.
Tietenberg, T. H. (1990) 'Economic Instruments for Environmental Regulation' in *Oxford Review of Economic Policy*, Vol. 6, No. 1, 17-31.
Tobey, J.A. and K.A. Reinert 1991. 'The Effects of Domestic Agricultural Policy Reform on Environmental Quality' in *The Journal of Agricultural Economics Research* Vol. 43, No. 2, 20-28.

Tsur, Y. 1993. 1993.'The Economics of Conjunctive Ground and Surface Water Irrigation Systems' Staff Paper P93-15, Department of Agricultural and Applied Economics, University of Minnesota.

UK Central Statistical Office. 1992. *Family Expenditure Survey - 1991*. London: HMSO.

UK Department of Social Security. 1994. *Household Below Average Income.* London: HMSO.

UK Department of the Environment. 1986. 'Nitrate in Water. Pollution' Pollution Paper 26, DOE, London.

UK Department of the Environment. 1991. *English House Condition Survey: 1986*. London: HMSO.

UK Department of the Environment. 1993. *Climate Change: Our National Programme for CO2 Emissions.* A Discussion Document. London: HMSO.

UK Department of the Environment. 1996. *English House Condition Survey: 1991*. London: HMSO.

UK Office of National Statistics .1996. *Family Spending: A Report on the Family Expenditure Survey 1995-96* . London: HMSO.

UK Office of National Statistics. 1996. *New Earnings Survey 1996: Analyses by Industry (Part C).* London: HMSO.

UK Office of Population Censuses and Surveys. 1996. *Mortality Statistics: Causes.* London: HMSO.

US Department of Agriculture. 1996. *Food Consumption, Prices, And Expenditures 1996*. Washington: USGPO.

Walls, Margaret and Jean Hanson. 1996. 'Distributional Impacts of an Environmental Tax Shift: The Case of Motor Vehicle Emission Taxes'. Resources for the Future Discussion Paper 96-11. Washington DC.

Ward, N. *et al*. 1993. 'Water Pollution From Agricultural Pesticides' Research Report Centre for Rural Economy, University of Newcastle-Upon-Tyne..

Weaver, Robert D., Jayson K. Harper and William J. Gillmeister. 1996. 'Efficacy of Standards vs. Incentives for Managing the Environmental Impacts of Agriculture.' in *Journal of Environmental Management* 46: 173-188.

Whitby, Mark. 1994. *Incentives for Countryside Management*. London: CAB International.

Whitelegg, John. 1993. *Transport for a Sustainable Future*. London: Belhaven Press.

World Health Organisation, Commission on Health and Environment. 1992. *Report of the Panel On Food and Agriculture*. Geneva: World Health Organisation.

World Wide Fund for Nature and Council for the Protection of Rural England. 1996. *Growing Greener: Sustainable Agriculture in the UK.* Godalming UK: WWF-UK.

World Wide Fund for Nature - International. 1995. *Pesticide Reduction : Economic Instruments.* World Wide Fund for Nature Discussion Paper. Gland, Switzerland. October.

Wu, Junjie, Mark L. Teague, Harry P. Mapp and Daniel J. Bernardo. 1995. 'An Empirical Analysis of the Relative Efficiency of Policy Instruments to Reduce Nitrate Water Pollution in the U.S. Southern High Plains' in *Canadian Journal of Agricultural Economics* Vol. 43, 403-420.