Sectoral dimensions of sustainable development: Energy and transport

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Introduction

Energy use is an important source of environmental pollution. Reducing emissions per unit of energy consumed and reducing the energy intensity of economic growth are therefore important components of sustainable development. Faced with this challenge, informed observers range from very pessimistic, to positively optimistic. The resulting debate has been acrimonious, passionate and not always illuminating, as the recent controversies surrounding Lomborg's book *The Skeptical Environmentalist* demonstrate.[1] Environmental pessimists argue that pollution is inexorably linked to fossil energy consumption, and that individually selfish countries see little benefit in reducing fossil energy intensity. Economic pessimists are prepared to accept that pollution could be reduced by intelligent tax policies, but that governments invariably choose very much less efficient policies that are likely to cost considerably more than the benefits. Thus Nordhaus and Boyer (1999) argue that the costs of the Kyoto Protocol are seven times the benefits, and almost eight times as high as a cost-effective strategy. The high and unjustified level of cost in turn will lead to the policies being abandoned, rather than being replaced by more efficient alternatives.

Optimists take up from where the pessimistic economists leave off. They accept that policies are frequently poorly designed, but take encouragement from a number of positive trends. They accept that tax or price-guided solutions to addressing the external costs of environmental pollution are normally superior to quantity controls or standards, and that policy makers have a preference for controls and standards. They note that predictions of the costs of imposing standards often turn out to be too high, as unforeseen innovations allow these standards to be met at modest and acceptable cost. Faced with a challenging quantitative target, rather than a tax that can be passed on to final consumers, technologists redirect and concentrate their creative efforts to deliver surprising improvements, while managers make cost-effective investments or change production practices.

A second defence of a more optimistic assessment is that where current solutions appear inefficient and poorly directed, there is an incentive first to improve the estimates of costs and benefits, and then to encourage benefit-cost tests of proposed remedies. In the UK, measures to address emissions have shifted from a requirement to install BAT (best available technology) to

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[1] Lomborg (2001), *Scientific American* January 2002, and the resulting debate partly reported on Lomborg's website http:/www.lomborg.org.

BATNEEC (best available technology not entailing excessive cost). Excessive cost logically means costs that outweigh the benefits, creating a demand for a quantification of the benefits of emissions reduction. This trend has been reinforced by a parallel trend towards electricity and gas liberalisation (often associated with privatisation) and hence a replacement of energy policy based on physical planning with the need for market-friendly alternatives, such as taxes or tradable permits. If taxes are to be set, or if quota prices feed through to final energy prices, then voting consumers will be able to judge the cost of meeting environmental objectives. That in turn is likely to force a reappraisal of whether the costs are justified, as well as stimulating developments to lower the cost of delivering those benefits. This should lead to a better balance between the costs of reductions and the benefits of improved environmental quality.

Optimists also expect that the cost of reducing emissions per unit of useful energy delivered may not be as high as feared, providing sufficient time is given for adapting the capital stock and developing new technologies. Long-run energy price elasticities are notoriously hard to estimate (Barker et al, 1995), but in some sectors (e.g. transport) could be above unity (in absolute terms). Emissions price elasticities are necessarily higher than energy price elasticities, and for many pollutants are considerably higher. Large reductions may be possible for modest tax increases, and the environmental taxes should allow other distortionary taxes (ultimately on labour supply) to be beneficially reduced.

The optimists are optimistic because they believe that improving the social efficiency of energy use (reducing emissions to cost-justified levels) requires reasonably well-defined policies, and that ultimately the political process will be forced to make more rather than less efficient policy choices. Economic pessimists believe that the difficulties of reaching efficient multilateral agreements make this unlikely. Both, however, agree that well-designed policies can substantially reduce the costs of meeting any given level of environmental improvement. This paper will therefore concentrate on identifying what such policies would look like, and how they may be quantified and implemented. We start with a brief review of the evolution of energy policy, and the determinants of energy use at the economy and sectoral level. This leads to the link between energy use and environmental pollution. The last part addresses the design of policy to achieve efficient energy use, and the extent to which countries are moving towards such policies.

The evolution of energy policy

Traditionally, energy policy was primarily concerned with security of supply and accessibility at acceptable prices to the population. These concerns remain, and are reflected in requirements to carry fuel stocks, provide adequate gas storage, and adequate electricity capacity. Universal service obligations and concerns over fuel poverty continue to influence energy taxation and pricing in often perverse ways. Security concerns were given fresh impetus by the 1973 oil embargo, that also precipitated the next major concern - that of the finiteness of energy resources. The Club of Rome's doom-laden predictions of imminent scarcity seemed to be supported by the sharp increase in the oil price (Meadows et. al, 1972). Natural resource economists appealed to Hotelling and argued that the scarcity rent of exhaustible resources would rise inexorably at the rate of interest, so that projections of future oil prices made in the 1970s reached alarming levels when projected to the end of the century. The United States embarked on a major research programme to develop alternative sources of energy, ranging from exploiting tar sands to exotic methods of developing electricity by photovoltaics, magnetohydrodynamics or fusion reactors. Several countries launched major nuclear programmes with France proceeding furthest down this route.

The shift from oil as the marginal fuel for electricity generation back to coal and the rapid penetration of gas depressed demand for oil and softened prices. Oil prices halved in 1986 when Saudi Arabia reasserted its position as swing producer and controller of world oil prices. In OECD countries as a whole, the share of oil in total primary energy supply (TPES) fell from 53% in 1973 to just under 41% in 2000. About three-fifths oil is now used in transport (overwhelmingly road transport) with only a fifth used in industry and a fifth in all other sectors (where two-thirds goes to residential use). In OECD-Europe the pattern of oil consumption is similar but the oil share in TPES has fallen even more rapidly from 54.5% in 1973 to 38.8% in 2000 (OECD, 2002).

Falling oil prices, rapid gas development and the delayed resumption of economic growth after the oil shocks and international financial crises of the 1970s and early 1980s raised new concerns. If oil did not appear to be running out, reserves of gas appeared large and growing, while coal appeared abundant and increasingly internationally traded, concerns about the environment rose rapidly up the political agenda. Environmental pollution was not new, and most industrial countries suffered heavy and damaging pollution from smoke until various Clean Air Laws were enacted. Controls on particulate emissions from power stations and the shift from coal to gas in the domestic sector led to dramatic environmental improvements in OECD countries, if not in the Soviet block. Concern shifted to acid rain, primarily from power stations, and smog, primarily from nitrogen oxides (NO_x) produced by road transport.

Transboundary pollutants, particularly sulphur dioxide, SO_2, were addressed in a series of international agreements and translated into national limits. As a result, sulphur dioxide emissions have been dramatically cut, partly by flue gas desulphurisation, and partly by the shift from coal to gas in electricity generation. Similarly, increasingly stringent tail pipe emissions limits have dramatically reduced pollution from road transport, to the point that some of the worst affected areas like Los Angeles now enjoy cleaner air despite massively greater traffic than in the early post-war years.

The health effects of air pollution have been carefully studied and quantified.[2] The challenge, which is increasingly accepted, is to encourage socially efficient levels of abatement. Whether this is best achieved by taxes or standards, or some combination, depends on the fuel, the use, and the type of user. The implication is that abatement measures must pass a social cost-benefit test, which requires an estimate of the monetary value of the damage caused.[3] Although internalising these pollution costs still presents an important challenge to the energy and transport sectors, and will be discussed below, concerns have shifted towards a more pervasive and difficult pollutant, carbon dioxide. The potential of increased levels of carbon dioxide, CO_2, to cause global warming has moved from scientific theory to widely accepted fact, reflected in the Kyoto Protocol to reduce CO_2 emissions in the near term, and to contemplate more dramatic reductions over the next 50 years.

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[2] For recent estimates, see the papers of the UNECE symposium *The measurement and economic valuation of the health effects of air pollution*, London, Feb. 2001 at http://www.unece.org/env/nebei.

[3] The EU has commissioned a series of studies to estimate the social costs of various emissions, and a recent set of marginal external cost estimates are provided in BeTa, the Benefits Table data base listed on the EC DG Environment web site.

Carbon dioxide emissions are far more intractable than other air pollutants, as it is difficult and extremely expensive to prevent or reduce CO_2 emissions from fuel combustion. The only practical methods for reducing CO_2 emissions are to shift to less carbon-intensive energy sources (and renewables, hydro,[4] wind, and nuclear have essentially zero emissions) and/or to reduce energy consumption.

Sustainable development

The watch word for energy policy is now not just security but sustainability, aptly described by the Brundtland Commissions definition of "development that meets the needs of the present without compromising the ability of future generation to meet their own needs." This concern with sustainability also reflects earlier concerns about the exhaustion of fossil fuels, and for transport fuels, oil exhaustion is probably a more imminent concern than excessive global warming. Coal reserves dwarf oil and gas reserves, and in that sense oil and gas are potentially smaller contributors to likely future greenhouse gas emissions, though none-the-less significant.

Old-style energy policy shared many of the characteristics of soviet planning, being quantity driven and not particularly susceptible to rational economic calculations. State-owned electricity industries built plant under central guidance, domestic coal was protected by a complex web of taxes and contracts with the electricity industry, gas was denied to electricity generators (as a noble fuel too valuable for simple steam raising), and in some countries district heating schemes were built by diktat or with massive subsidies. All this started to change following Alfred Kahn's successful attack on regulation in the airline industry and the liberalisation of traditional utilities, first telecoms, and then gas and electricity. Privatisation in Europe, unbundling, and increasing attempts to use competitive markets rather than regulation for setting prices, unleashed dramatic changes for the energy sector, and forced a reappraisal of energy policy.

This became clear in Britain soon after the electricity supply industry was restructured and privatised in 1989-90. To ensure a satisfactory sale and to provide a smooth transition to an unregulated electricity wholesale market, the Government put in place three-year contracts for the purchase of coal and the sale of electricity. As the end of these contracts approached, it became clear to an increasing number of observers (and finally to the Government) that there would be a dramatic decrease in the price and quantity of British coal that would be purchased in future, partly because imported coal was cheaper, but mainly because the "dash for gas" was well under way. The Government was criticised for not acting to protect the coal industry (i.e. the powerful miners) and for lacking any energy policy. Energy policy in Britain, as in most countries, is almost by definition politicised, for to leave the choice of fuel to an undistorted market is thought to characterise the lack of an energy policy. The Government felt the need to defend its unprecedently non-interventionist stance after the collapse of the coal market in 1992 by arguing that 'The aim of the Government's energy policy is to ensure secure, diverse and sustainable supplies of energy in the forms that people and businesses want, and at competitive prices.'(DTI, 1993, p12). 'The Government's energy policy therefore centres on the creation of competitive markets.'(*ibid*, p3).

[4] Large-scale hydro can, by inundating plant matter, lead to decomposition and the release of methane, a far more potent greenhouse gas than CO_2. Renewables absorb CO_2 from the air and release it again when burned, so produce no net emissions provided they are replaced and not mined.

Concerns over the possible tension between liberalisation and sustainability (the new concern of energy policy) have been expressed in various IEA reports (e.g. IEA, 1998) as well as by the European Commission. Thus one of the criticisms levelled at electricity liberalisation is that "if the internal market causes electricity and gas prices to fall, this in turn would probably lead to an increase in consumption" causing an increase in pollutant emissions and hindering attempts to honour commitments made in Kyoto. (OJ, 2002; 6.4.9.2)

Economists, and increasingly public bodies advocating more market-friendly policies and interventions, argue that there is no inevitable tension between liberalisation and sustainability, providing that market prices are corrected by taxes to reflect all external costs, in this case those that cause social and environmental damage. Thus the British Government, in its report *Sustainable Development: the UK Strategy* (HMSO 1994) interpreted sustainability for transport as requiring that "users pay the full social and environmental cost of their transport decisions, so improving the overall efficiency of these transport decisions for the economy as a whole and bringing environmental benefits." (HMSO, 1994, p6, 169). The same holds not just for transport decisions, but any decisions involving fuel.

If users must pay the full social cost of their decisions, society will be compensated for the environmental damage done, and can use the funds to make other environmental improvements, or to accumulate more physical and human capital, making the next generation richer and better able to address environmental issues. The benchmark that users should pay the full social and environmental cost is therefore central to the idea of a decentralised and non-coercive approach to dealing with environmental problems. It is completely compatible with liberalised energy markets, providing that the social costs are reflected in market prices, normally best done by corrective taxes.

At this point it is useful to distinguish between stock and flow pollutants. Flow pollutants cause damage while they are being produced, and the damage ceases when emissions stop. The larger part of the social and environmental cost of SO_2 and NO_x is the health damage caused by inhalation. Reducing emissions of these pollutants has an immediately beneficial effect on air quality as it affects health. Stock pollutants in contrast add to the stock of the pollutant, and it is the size of this stock rather than the rate of addition that causes the damage. Acid rain damages the eco-system by increasing the acidity of the environment, while CO_2 emissions add to the total stock of atmospheric CO_2 that is the main cause of global warming.

Flow pollutants are in principle easier to price than stock pollutants, as we only need to know their instantaneous rate of damage, normally through dose-response relationships, in ways illustrated below. In contrast, the damage done by stock pollutants endures over time, and would need an accounting of the damage done at each future period (that will likely depend on future emissions as well) and also on the discount rate. Future emissions are hard to predict and will likely depend on future technical progress, while the choice of discount rate is also controversial.

The practical implications of this is that some energy-related pollutants lend themselves better than others to market solutions such as taxes (though standards may also be important where measuring emissions is difficult). Carbon dioxide is an interesting case, for on the face of it, it is the pollutant that best fits the need for a tax solution, as the damage done is directly proportional to the carbon content of the fuel to be burned, and does not depend on where or how that fuel is burned. Contrast that with NO_x, whose emissions depend on the temperature at which the fuel is burned, how it is burned, whether it is subject to tail-pipe clean up, and whose damage depends on when and where it is released. Nevertheless, the correct or appropriate carbon tax to levy depends on reaching agreement across country borders, as well as on future CO_2 emissions. Not surprisingly, the range of estimates of the appropriate carbon tax is embarrassingly wide.

That suggests breaking the question of how to identify the right energy policy for sustainable development into a number of separate components. The first question to examine is what determines the demand for energy (and of different fuels), as other things equal, a reduction in energy used will reduce the problem. If it appears that energy intensities can be significantly reduced, then the gains from intelligent policy are likely to be large. The second stage is whether and to what extent it is possible to reduce pollutants per unit of energy, since these are the ultimate source of the damage. Even if we cannot reduce energy use beyond a certain point, if it is possible to reduce the pollution per unit of energy, then it should be possible to go considerably further towards sustainable development.

Once the potential for improvement has been established, the next step is to encourage socially efficient choices (of energy and emissions abatement). That requires three conditions - first, that decision makes confront the right relative prices; second, that decision makers can identify and access the efficient choice; and finally, that technical progress is directed to delivering the best future choices. Getting the prices right means correcting for the social costs of the choices, as well as avoiding tax-induced or other distortions. Ensuring that decision makers choose efficiently if prices are right is more complex. It requires that agents are well-informed, rational, and have the right incentives to make efficient choices. Failures here may be identified by benchmarking companies (or other decision-making units in the public or domestic sector) against best practice, and may be explained by a lack of information or a lack of incentive. Agents may not be aware of the energy efficiency of the products they buy, or of the full costs of their operation. If they delegate decisions, their agents may not have sufficiently strong incentives for efficient choices, particularly where the benefits are hard to measure and occur with a lag.

A large part of the sustainability policy agenda is directed to providing information, benchmarking, auditing, and stimulating research in promising directions. However, if the prices facing decision makers are systematically distorted away from their socially efficient level, much of this effort will be undermined, including the incentives to develop appropriate new technologies. The main thrust of this paper will therefore be on the rationality of the economic signals facing agents when making energy and pollution abatement decisions.

Energy use and growth

Figure 1 graphs the relationship between energy use (measured in thousands of tonnes of oil equivalent, ktoe) and GDP at constant US$ 1995, using World Bank data.[5] The approach taken in this paper is to concentrate on certain countries and groups of countries that span the range of UNECE countries (with the intentional exception of Japan). The main emphasis is on Europe, where we distinguish the EU countries as a group as well as some member states, Norway, several of the accession transitional countries (Hungary, Poland, and sometimes also the Czech Republic), Russia, and the USA. Each point in the graph is the energy use in a particular year for that country or group, and the graphs therefore show the evolution over time of the relationship between energy use and GDP. The graph is double logarithmic and the slopes of the graphs give

[5] The data are available in electronic form in the World Bank's World Development Indicators database, summaries of which are published annually in the *World Development Report*.

the elasticity of energy use with GDP. The line of 0.18kg oil equivalent per US$1995 (which is the average over the period for the EU) indicates a unit elastic relationship in which a 1% increase in GDP would lead to a 1% increase in energy consumption.

It is clear that GDP is a main determinant of energy use, but it is also clear that some countries, notably transition countries and the USA, are notably more energy intensive than the EU countries. It is less obvious from the graph that in most countries energy intensity (toe/$) has been falling slowly (by 1.2% p.a. in the EU and 1.9% p.a. in the US), while the variation in energy intensity across the EU and also including the US has also been decreasing, indicating some convergence to a common energy intensity.

Figure 2 shows the relationship between energy use and real GDP at international prices or Purchasing Power Parity, PPP, (taken from the Penn World tables). This is arguably a better measure of relative standards of living, and corrects for the lower cost of non-tradable goods in poorer countries (and some of the distortions caused by taxes and tariffs). This graphs covers a longer period than Fig. 1 (from 1960 for most countries, compared to from 1972 in Fig. 1). For almost every country except Russia the ratio of RGDP to $GDP is roughly constant for each country from 1972-99, though the ratio varies widely across countries.[6] The result is to move transition countries closer to the unit elastic line drawn through the EU average intensity.

Figures 3 and 4 show energy use per head versus the two measures of income per head, this time on arithmetic scales. Again, the longer graphs in Figure 4 reflect the longer time period covered (except for the transition countries). Lines of constant slope through the origin represent constant energy intensities (kgoe/$) and the graph for Russia (as in Figures 1 and 2) shows decreasing energy use with a fall in income over time, in contrast to all other countries where income per head typically increases. These two figures show the dramatic fall in energy use in the USA from 1978 to 1983 (the second oil shock and recession).

The difference between the two measures for the transition countries is clearly dramatic: at market exchange rates, Russia is ten times as energy intensive as the EU, but only three times at PPP. Similarly, the energy intensities of the Czech and Slovak Republics and Romania are more than three times as high at market exchange rates than at PPP, and those of Hungary and Poland are more than twice as high. This raises the obvious question of which measure is the better one for understanding energy demand and, more importantly, the potentials for reducing energy intensity. Eastern Europe was notorious for subsidising energy (and other consumer goods, many of which were rationed), and collecting taxes directly from enterprises rather than from consumers or workers. Consumer purchasing power was therefore greater than appeared at market exchange rates (which were in any case heavily distorted by the Comecon system of trade), and by that measure real income was higher than at market exchange rates.

The important question to which we turn next is whether the transition to a market economy facing ultimately less distorted prices will reduce the energy intensity, however measured. Over time (and the direction of movement may not always be apparent in the figures) the transition economies do seem to be moving towards the line of average energy intensity, supporting this hypothesis. As inefficient and resource-intensive heavy industry has been confronted with world market prices, so much of it has contracted, exited, or improved its efficiency. Nevertheless, the process of moving *effective* energy prices (that buyers actually pay)

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[6] For eastern European countries RGDP is between 2 and 4 times $GDP, for the US is defined as unity, while EU countries range between 75% and 123% (Portugal).

towards world market levels has been painfully slow in many countries, so this adjustment is likely to take some considerable time.

To summarise briefly on the relevance of measuring income levels at PPP, where there are larger differences than might be expected purely in terms of per capita income levels, the reason is likely to lie with highly distorted price structures, which in many cases are part of the reason for profligate and inefficient energy use. Using PPP exchange rates to deflate local energy prices may give a better impression of how expensive energy seems to domestic consumers, but is not an adequate reason for subsidising energy to those consumers, given that (most) energy is internationally tradable, and should therefore have its price linked to the international price at the market exchange rate. For the rest of the discussion we shall therefore only use market exchange rates when making comparisons.

Sectoral energy use

In OECD Europe in 2000, 35% of energy was consumed by industry, 20% in transport, 25% in the residential sector, with 20% in other sectors (OECD, 2002).[7] In the US, the shares were more equal, at 27%, 26%, 24% and 23%. In both cases, the absolute amounts of energy (tonnes oil equivalent) consumed in industry have remained remarkably stable since 1973, but transport use has grown rapidly (nearly doubling), and other sectors have increased by about 20%. The share in industry has therefore fallen over time. What is perhaps rather surprising is that the energy intensity of industry is very similar to the economy as a whole (total energy, including inputs into electricity consumed, per $ value added).[8]

Table 2 shows that industrial energy intensity has fallen by nearly 50% in EU countries since 1971, and by 64% in the US, considerably faster than economy-wide energy intensities, which fell by 28% from 1972-99 in the EU and by 40% in the US. Estimating industrial energy intensities in transition countries is more difficult, particularly given the different system of national accounting and the distorted price structure, but there is some evidence that again the sector energy intensities are similar to the economy intensities. That in turn implies that energy intensities are far higher than in market economies, reflecting inefficient resource use associated with central planning and distorted prices.

The rapid growth in transport energy use is primarily the result in the rapid growth in vehicle km travelled (VKT). In Britain, energy use per VKT has fallen from 104gm oil equivalent/km in 1980 to 89gmoe/km in 2000, or an increase in energy efficiency of 15%.[9] Looking across countries, fuel efficiency varies - in large part because of variations in transport fuel taxes and hence prices. Thus US fuel consumption per km is 40% higher than in the EU, and US transport fuel taxes and hence prices are significantly lower than EU taxes.

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[7] This includes the fuel used for generating electricity which is consumed by the end-use sectors.

[8] Estimating sectoral energy intensities requires finding comparable coverages of sectors for both energy and value added, which is time consuming. The estimates here rely on World Bank data for the share of industry in GDP and OECD energy balance data for energy consumption by sector, and the coverages may not be exactly the same.

[9] Calculated from data for total vehicle km and total energy use in Department for Transport *Transport Statistics Great Britain 2002* and earlier years. The rate of increase of efficiency from 1989-81 is rapid, and then stabilises until 1985 before continuing to improve, so the results are sensitive to the start date chosen.

The effect of price on energy use

The natural explanation for the considerable variation in energy intensity across countries and the relatively slow decrease in energy intensity with time is that prices vary more across countries than over time. In addition, energy use responds slowly to price changes, as the energy-using capital stock takes time to adjust to different energy prices - decades in the case of power stations and buildings, and maybe a decade for vehicles and machinery. Very simple cross-country econometrics suggests an economy-wide energy price elasticity of about -1 (Hoeller and Wallin, 1991; Barker, Ekins and Johnstone, 1995; Cooper et. al., 1999). More detailed studies of particular sectors suggest a similar aggregate figure, though the sector levels may be higher or lower. Estimates of gasoline price elasticities in transport bracket -1 from cross-section studies, but up to -2.3 for the long-run elasticity estimated from time series analysis (Franzén and Sterner, 1995). Industrial energy price elasticities may be lower (-0.3 to -0.5), but output elasticities are also below unity, suggesting a trend growth of energy efficiency even without price changes (Vouyoukas, 1995). In all cases the short-run elasticities are much lower than in the long run, typically around -0.1 to -0.2.

Fig. 5 illustrates the cross-country relationship between average energy price (US$/toe, weighted by fuel share and sector) and average economy-wide energy intensity, for OECD countries from 1993-99, using data from OECD (2002).[10] The constant elastic regression line is fitted for non-transition countries, and suggests that the energy intensity of transition countries is higher than expected given their energy prices. That is consistent with other factors (e.g. central planning) influencing energy use. Note that the maintained assumption in fig. 5 is that GDP elasticity of energy use is unity. Once again, the cross-section price elasticity is –1.0 with a standard error of 0.14, consistent with the other evidence.

This economy-wide elasticity of about -1 is probably picking up other policies as well as those associated with price changes, for when oil prices rose sharply, countries often imposed additional incentives to reduce oil imports and energy consumption. Thus, prompted by the oil price shocks, the US imposed CAFE requirements on motor manufacturers to meet fleet average fuel efficiency standards. It could also be argued that these associated policies were primarily aimed at encouraging the capital stock to adjust more rapidly to future expected prices, and is therefore one of the routes by which price changes feed through into final energy choices.

The implication is that energy prices and possibly other complimentary policies can have powerful effects on energy use in total (and an even more powerful effect on the choice of fuels in some sectors, such as heating, steam raising, electricity generation, and the choice between gasoline and diesel for vehicles). To put this into perspective, if the price elasticity is -1, and the GDP energy elasticity is 1 (i.e. energy intensity is invariant to income levels), and GDP grows at 3% p.a., then energy use would not increase if prices rose at 3% p.a. in real terms (i.e. relative to all other prices). This would mean a price increase of 80% over 20 years. Not surprisingly, most attempts to decouple energy use from GDP growth involve steady increases in real energy prices. For oil and eventually gas, such price rises are consistent with resource depletion (and were the default price forecast in the 1970s), but for coal and tar sands, stocks are too large for resource depletion alone to drive steady price increases. Some have therefore argued that energy taxes should be steadily increased to depress the otherwise inexorable growth of energy use with

[10] Average prices and energy intensities are used as energy use responds slowly to price changes.

economic growth. Extremists have gone further and argued that economic growth itself should be curtailed.

Simple energy taxes alone do not, however, make sense, as it is the harmful emissions that are the reason for action, not energy consumption per se. The time has come to switch attention from the determinants of energy use to those of the resulting environmental pollutants.

Policy to increase the efficiency of energy use

Sustainable energy use requires that market prices are corrected by taxes (or their equivalent) to reflect all external costs. How much the current generation bequeaths to the future can then be determined by the amount of capital accumulated, where capital includes not only physical capital, but also human, social, environmental, and natural resource capital. The revenue from these corrective taxes can be used to purchase not just further reductions in these pollutants, but other services that may yield higher increases in welfare. If emissions of sulphur dioxide reduce life expectancy, then sulphur emission taxes may be able to buy more quality-adjusted years by transfers to the health service than being spent on increasingly expensive sulphur abatement. More generally, charging according to damage done, and allocating the revenues to where the greatest benefit occurs, is more efficient than forcing pre-specified levels of abatement.

The four main pollutants associated with fossil energy consumption are particulates (black smoke) from incomplete combustion, the acid rain precursors sulphur dioxide and nitrogen oxides, and green house gases, particularly carbon dioxide. We first discuss their relation to energy use, and the extent to which they can be or have been decoupled from energy use, and then discuss the design of appropriate ways of confronting energy users with their social costs.

Carbon dioxide emissions

Carbon dioxide, CO_2, is released in direct proportion to the carbon content of the fuel, and is the main cause of climate change (global warming). Fuels vary in the carbon content per unit of useful energy, measured by e.g. tonnes carbon per tonne oil equivalent (tC/toe), or tonnes CO_2/toe as in fig. 6.[11] Thus bituminous coal has 1.1 tC/toe, gasoline 0.8tC/toe, HFO 0.88tC/toe, natural gas 0.64tC/toe, while nuclear energy, renewables and hydro-electricity have zero values.[12] The wide range of values in fig. 6 reflects at one extreme Norway's heavy dependence on hydro-electricity (also used for domestic heating) and at the other the heavily coal-dependent country Poland. The rapid decrease in CO_2 intensity in the UK reflects the switch from coal and oil to natural gas and the development of nuclear energy. The larger fall in France reflects the more complete penetration of nuclear power in electricity that has moved France from near Poland's intensity to that of Norway.

Clearly, then, CO_2 intensities can be reduced by fuel switching, particularly to non-carbon based electricity. Norway and France demonstrate that the gains from switching are limited by the overwhelming dependence of transport on oil.

Emissions of other pollutants

[11] The ratio of tonnes CO_2 to tonnes C is 44:12, or 3.67:1.

[12] Taken from IEA (2002b). Note that tC/tonne of fuel may be quite different, e.g. only 0.67 tC/tonne of bituminous coal.

Climate change *policy*, if not concern, is relatively recent, and the Kyoto Protocol has still not been signed by key players such as the US. In order to judge how effective environmental policy can be, it is useful to look back at the dramatic successes of earlier policies towards other air pollutants. In each case, once the damage had been recognised, local action was taken. Where the damage spilled over national frontiers, a surprising degree of international agreement and action followed. Economists have criticised some of these agreements and legislation as inefficient, or unjustified on social narrow cost-benefit terms (see e.g. Newbery, 1990; Crandall et. al, 1986), but judged purely in quantitative terms their impact has been impressive. The evidence below is taken mainly from British sources, but the same results could be found in most developed countries.

For most pollutants, policy acts at two levels. Typically the source of the problem is addressed by imposing emission limits, at the national level and/or at the source level, for total emissions and/or emissions per km travelled or per kWh generated. Early legislation concentrated more on controlling emissions, while later legislation addresses the resulting air quality standards. Thus in the EU, the Framework Directive 96/62/EC requires a preliminary assessment of air quality by certain dates for each pollutant, and hence to be in a position to detect exceedences.

Particulates

The first major environmental pollutant to attract attention and legislation was smoke from burning coal. Londoners in the twelfth century complained about the noxious fumes from burning sea coal, and the corrosive effects of sulphur dioxide, SO_2, dissolved in rain has been well understood for at least a century. In Britain, policies to address the harmful effects started with the Smoke Abatement Acts of 1853-56, and via various other measures to the landmark Clean Air Act of 1956, followed by a second Clean Air Act in 1968. Concentrations of SO_2 in London fell from $900\mu g/m^3$ in 1850 to $25\mu g/m^3$ in 2000.[13] The proximate cause for this later legislation was the very obvious health hazards associated with the unregulated burning of coal, and in particular the large number of people, estimated at 4,000, who died in the great London smog of December 1952. The incomplete combustion of coal (and oil) produces fine suspended particulates that are damaging to health, and increase mortality and morbidity. The severity of the risk increases with the concentration of fine particulate matter, which is measured by the concentration of particles of less than 10 microns (PM_{10}).

The combined effect of legislation, which prompted the development of smokeless fuels to replace coal in designated areas, and the gradual replacement of open coal fires by central heating (now mainly gas-fired) in domestic use, dramatically reduced PM_{10} emissions, as shown in fig. 7. Total emissions fell by 46% in the decade from 1970-1980, while emissions from the domestic sector (that are the most damaging as they come from low level sources in densely populated areas) fell by 58% (Department of the Environment, 1995).[14] The rate of decrease continued, with domestic emissions falling 49% in the decade 1980-1990 and a further 47% from 1990-2000. Over the whole period 1970-2000 total emissions fell by 74%. Domestic emissions now only account 20% of a much smaller total, compared to 42% in 1970. Similarly, power

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[13] See Lomborg (2001, p165) for sources, which for 1850 are derived from coal imports.

[14] Particulate emissions before 1980 are measured by black smoke, which correlates closely with PM_{10}.

station emissions have fallen dramatically, by 81% per kWh from 1970 and by 76% in the period 1990-99 (primarily as a result of the switch from coal to gas), while emissions per km travelled by road transport has fallen 73% since 1970 and by 57% since 1990.

Records of emissions and concentrations in other countries typically cover shorter time periods.[15] Thus in Germany, if the data are to be believed,[16] total emissions fell from 2059 kt ('000 tonnes) in 1990 to 864 kt in 1992, or a drop of 58% in just two years.

Sulphur Dioxide

Sulphur dioxide, primarily from burning coal and oil, is a prime contributor to acid rain, and when released from high stacks can travel considerable distances as an aerosol, causing damage downwind and in other countries. The fact that the damage might be done to other countries and could not therefore be addressed by national action alone, was a prime factor leading to international agreements and protocols. Different countries responded to different facets of the pollution problem. The Scandinavian countries were troubled by the death and disappearance of fish from lakes and rivers. Germans worried about forest die-back. Glasnost revealed the full extent of the environmental disasters in East Europe, and provided the focus for local hostility to the environmental insensitivity of central planning.

The debate on acid rain and on appropriate responses has been conducted in two different forums. The initial pressures came from the UNECE Convention on Long-Range Transboundary Air Pollution (LRTAP) that has 34 members from Europe and North America. Much of the pressure here was exerted by the Scandinavian countries and Canada, who are both large net importers of acid rain because of their unfortunate downwind location. In 1982, Norway and Sweden pressed for the signatories to reduce SO_2 emissions to 30% below 1980 levels by 1993. This led to an informal '30% Club' founded in Ottawa in March 1984, and, in July 1985 21 countries, but not including the US and the UK, signed a protocol at the third meeting of the UNECE LRTAP Convention in Helsinki.

Whereas the Scandinavians were initially primarily concerned with the acidification of lakes and streams and consequent loss of fish, West Germans were worried about the impact their own industry was having on the environment, concerns which were reflected in the growing political power of Green parties in the early 1980s. An emotive campaign in 1982 drew attention to the problem of *Waldsterben* or forest death, in which official estimates showed that over half the forest area had suffered damage, attributed to acid rain. For a variety of political reasons described in more detail in Berkhout *et al* (1989), a Large Combustion Plant Ordinance (Grossfeuerungsanlagen-Verordnung or GFAVo) was enacted in June 1983, under which flue gas desulphurization equipment would be fitted to 37 GW of coal fired power stations, and to the early closure of a further 12 GW. Not surprisingly, industry protested that the costs of this programme, which were to be borne by electricity consumers, would harm West Germany's competitive position in international markets, and this led the government to press for similar standards being adopted for the whole of Europe. The European Commission proposed a Large Combustion Plant (LCP) Directive based on the GAFVo in December 1983, calling for a cut in

[15] See e.g. http://europa.eu.int/comm/environment/air and the detailed studies on individual pollutants accessible from there.

[16] See table at p65 of http://europa.eu.int/comm/environment/air/pdf/pp_pm.pdf. Presumably the dramatic fall from 1990 to 1991 reflects the rapid restructuring of East Germany after reunification.

SO_2 emissions by 1995 by 60%, to 40% of their 1980 level. After much debate, described in Skea (1988), the UK finally agreed to reduce SO_2 emissions from existing large plant to 20% below its 1980 level by 1993, by 40% below by 1998 and to 60% below by 2003, and nitrogen oxides (on the same basis) by 15% to 1993 and 30% by 1998. The Directive also provides for stringent emissions standards for new large combustion plants which the UK accepted (Department of the Environment, 1988). In November 1988, the UK Environment Minister signed a UN protocol in Sofia committing the UK and most leading industrial countries to freeze the level of nitrogen oxides at 1987 levels until 1994 and by 1996 to agree to further reductions based on critical levels. In due course the Second Sulphur Protocol was signed and has had a significant effect on efforts to reduce sulphur dioxide emissions.

Britain reduced SO_2 emissions by 82% between 1970 and 2000, and although the decadal decrease was only 24% until 1990, from then the impact of international agreements (and their translation into national limits) has been dramatic, with an overall decrease of 69% from 1990-2000, and a 76% decrease in SO_2/kWh in power production (which in 1990 accounted for just under 80% of total emissions). Britain emitted only 76% of the 1998 LCP ceiling by the deadline date, and in 1999 was only emitting 87% of the substantially more stringent 2003 limit.

Detailed data on emissions from European countries is available from the European Monitoring and Evaluation Programme (EMEP). This was set up in 1978 to monitor the movement of pollutants, and to determine where the deposition of pollutants released from each source occur. For the EU, emissions fell 20% between 1980 and 1990, and were forecast to fall by a further 60%-91% between 1990 and 2010. The declines in other European countries (which accounted for as much as the EU) was similarly forecast between 29% and 86% over the same period.[17] Among the transition countries SO_2 fell by more than 35% in the CIS countries (but as energy use fell by 28%, emissions per toe fell by 28%), and by 25% in Central Europe, Baltic states and South Eastern Europe, the latter as a result of lower emissions per toe (EBRD, 2001, p93).

Nitrogen oxides

Nitrogen oxides, NO_x, are produced from the air involved in combustion processes, rather than in pollutants contained in the fuel itself, as with SO_2. As with SO_2, country level data on NO_x emissions are available from the European Monitoring and Evaluation Programme. In Europe as a whole in 1994, 57% of total NO_x emissions come from transport, and 39% from stationary combustion sources (CORINAIR, 1995). In the EU in 1995, 62% came from transport, and 34% from stationary sources. In the US, the original impetus to address the problem was the deteriorating air quality in urban areas like Los Angeles and Washington DC, where photochemical smog led to high levels of ozone. This was traced to exhaust emissions of hydrocarbons and nitrogen oxides and California lead the way in introducing successively tighter emissions controls on vehicles. These have been picked up in other countries in emissions standards for vehicles (where international trade is a powerful mechanism for standardisation). In Europe, acidification was again one of the major forces for policy change. (In terms of acidification, NO_x is counted as 70% as damaging per tonne as SO_2.) As with SO_2, the approach was to impose limits on emissions from Large Combustion Plants (and on emissions in gm per km for vehicles). The EC Large Combustion Plants Directive 88/609/EC required that emissions

[17] See p9 of http://europa.eu.int/comm/environment/air/pdf/pp_so2.pdf.

in 1998 be 40% below those of 1980.

Again the evidence from Britain is instructive, and covers a longer time period than most other countries. Total emissions fell by 40% from 1970-2000, but as they rose by 10% between 1970 and 1990, the subsequent fall has been a more dramatic 45% in the single decade 1990-2000. Emissions from road transport per km travelled fell by 58% from 1990-2000, and from power stations (which accounted for 28% of the total in 1990) fell by 60% per kWh over the same period. By 1998 Britain was producing only 55% of the LCP target for that year. Thus Britain demonstrates that once the problem was recognised as important, and was systematically addressed by standards (for vehicles) and emission limits (for large plants), dramatic reductions were achieved. In the case of vehicles, as the proportion of vehicles meeting the more recent tighter limits increases, so emissions per km and in total are forecast to continue falling.

Designing efficient energy pollution policies

At one level, CO_2 is conceptually the simplest pollutant, for the damage done does not depend on where the emission occurs, and is directly proportional to the carbon content of the fuel. A fuel tax per tonne of carbon is therefore the logical instrument, and one that has been adopted by a number of Scandinavian countries (none of which has an indigenous coal industry to protect). The obvious problem is that the benefits of reducing CO_2 emissions in one country are overwhelmingly captured by other countries, so the incentive for any one country to unilaterally tax CO_2 is minimal.

The second problem is that it is difficult to quantify the present discounted total global benefits to reducing emissions of CO_2 now. There are long lags between emissions and their impacts on global temperatures, and then on ecological, climate, and sea-level changes. These impacts have very different effects on different countries, with the adverse impacts falling disproportionately on poorer tropical developing countries. Indeed, some calculations suggest that for modest global warming, some richer countries might even benefit, or at least lose only to a limited extent (IPCC, 2001). The damage done will depend on the direct costs of the damage and the costs taken to avoid damages (better sea-level defences, etc.), both of which will depend on the future state of technology. Even the discount rate to use is controversial, as normal commercial discount rates make damages a century hence of negligible present value. Thus the present value of $1 million in 100 years time at 5% p.a. discount rate is only $7,600. Lower discount rates can be defended if future income levels are substantially lower than at present, but most forecasts do not predict that outcome. Finally, the cost of reducing CO_2 emissions in the future depends on technical progress, and is also uncertain. That affects the optimum (cost-minimising) path of CO_2 reductions, and hence the time path (and present level of) carbon taxes.

Uncertainty is no excuse for inaction (though it may be a cause for delaying irreversible and costly actions if information can be improved), and considerable effort is being deployed to solve for the global optimum climate change policy and the implied level of taxes. In practical political terms it is most improbable that a uniform global level of carbon taxes could be agreed, but the Kyoto Protocol suggests a feasible and equivalent alternative. If global limits on CO_2 emissions can be agreed and quotas allocated to countries, and if these carbon quotas are freely tradable, then a global carbon permit price should emerge, in a competitive trading environment.[18] If energy users had to obtain these CO_2 permits in proportion to fuel purchased,

[18] The qualification is important for if major CO_2 emitting countries like the US (with 24% of the

then the external cost (at least of global warming) would be efficiently internalised.

The main objection to the presently formulated Kyoto Protocol is that it does not foster global (as opposed to regionally restricted) carbon trading, as most developing countries are exempt. (Another major objection is that by itself the Protocol will only delay century-hence levels of atmospheric CO_2 by about 6 years, and so a whole sequence of future CO_2 restrictions will be needed.) As a result the costs of compliance could be many times as high as the least cost global trading solution (eight times, according to an estimate of Nordhaus and Boyer, 1999).[19] It is easy to see why. Fig. 8 shows the marginal cost of reducing CO_2 emissions by 1 tonne in two different countries, at the pre-trade allowed level of emissions, OQ. The extra cost of reducing emissions in country A is QA, considerably higher than that in country B, QB. If country A could purchase a quota from country B, the cost saving would be (approximately) QA-QB, which could be many times the value QB (and also many times the equilibrium price with free trade between these two countries, QE).

Global carbon permit trading combined with country quotas for every country (and monitoring for compliance) would create the right conditions for selecting the efficient choices from those available, but would not guarantee that the right technologies would be produced. RD&D is a quasi-public good (even with good patent protection), and the learning spill-overs from developing and producing new carbon-saving technologies are likely to be large. That is a prime argument for subsidising low-carbon technologies, and an argument for international agreements to fund such research and set targets for their introduction, in order to stimulate their development and deployment. Fortunately, it seems somewhat easier to persuade individual governments to support local initiatives than economists, fearful of free-riding, might have expected. Support for low-carbon energy is even more important if developing countries lack economic incentives to reduce CO_2 emissions, for if these technologies become competitive against traded fossil fuel, then these countries would have an incentive to use these rather than carbon-intensive fuels.

If governments appear to be taking the development of low-carbon energy substitutes reasonably seriously, the same cannot yet be said for pricing carbon, at least in most countries. Britain, for example, has introduced a Climate Change Levy which is a pure energy tax (with exemptions for renewables, good quality CHP, but not for nuclear electricity). Households, whose energy is already subsidised though a reduction in VAT of 12.5%, are exempt, although improving household energy efficiency is probably one of the least-cost ways of lowering overall energy use. A more logical approach would be for each government to levy the carbon tax needed to meet its Kyoto commitment (or, on an optimistic view, at the level of the future equilibrium price of traded carbon permits). If carbon permits become tradable, then the

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world total) Russia, (with a likely large trade surplus of permits) or China were to use their market power, the internal and world prices would not be equilibrated.

[19] Nordhaus's model suffers from serious methodological problems (Barker, 1996); and the costs of mitigation depend sensitively on how the tax revenues are recycled (and the size of the "double dividend"), how far carbon reductions lead to additional reductions of other pollutants, as well as the extent of trade. Barker and Ekins (2003) compare the range of cost estimates presented in Watson (2001) for differing degrees of trading. The ratio of the costs of mitigation with no trade compared to global trading range from 2.3 to 22, with most estimates between 5-7. The ratio of the costs with trade limited to Annex I countries to global trading cluster around 3. All these figures are affected by revenue recyling and ancillary benefits.

government can replace the carbon tax by the requirement to buy carbon permits, ideally at the same rate as the tax.

Carbon dioxide emissions are not the only reason for taxing fuels, but if we inquire how energy is actually taxed, it is hard to relate taxes to potential damage, as the next section demonstrates.

The current pattern of energy taxes and subsidies

Different fuels are taxed at very different rates within almost all OECD countries, and the same fuel is taxed at very different rates across the OECD. Fig.9 gives the average mineral oil tax rates for 1997 across EU countries, defined as total tax revenue (excluding VAT) divided by final consumption of oil products. To gain a rough sense of the tax rates, the pre-tax price of oil products in 1997 were probably on average Euro 100/tonne oil equivalent, so the taxes as a percentage of product prices were high (substantially greater than 100%). In contrast, coal is normally untaxed (except in Denmark and Finland), as is gas for industry (with a few more exceptions).

Hydrocarbon taxes are also fiscally important. On average they contribute 2% of GDP to the budget (as shown on the right hand scale of fig. 9), or about 5% of tax revenue. The UK stands out as having heavy oil taxes, primarily but not solely arising from the heavy taxation of road fuels. The ratio of hydrocarbon taxes to total UK government revenue has risen from 4.5% in 1989 to 6.7% in 1999, and the real tax receipts have grown at 6.2% p.a. over this decade. More to the point, hydrocarbon taxes account for a significant share of indirect taxes - in the UK 20% of all indirect tax revenue (including VAT), and 46% of indirect taxes if VAT and import duties are excluded.

The variation of EU tax rates (as a percentage of the pre-tax price, and again excluding VAT) for different fuels for the industrial sector is shown in fig. 10. Light fuel oil (LFO) stands out as heavily taxed in some countries, notably Italy, Portugal and Greece, presumably where there are difficulties in preventing tax evasion on the even more heavily taxed road diesel fuel, for which kerosene can readily be substituted. Heavy fuel oil (HFO) is relatively heavily taxed in Sweden and Austria, while Denmark appears to have the most uniform tax system across fuels, as the base is primarily carbon content.

Fig. 11 shows the taxes on fuel consumed in the EU domestic sector (excluding road fuel, which is shown in fig. 10). LFO is primarily used for central heating, as is gas, but they are taxed at very different rates (except in Denmark), again probably to prevent road fuel tax evasion. The variation across countries is considerably larger than for industrial use, as one might expect on efficiency grounds. The average tax rates are typically higher than for industry, again as expected.

Fig. 12 completes the picture by comparing taxes on road fuel across the EU. The tax *rates* were more than 250% of the pre-tax price on average, and over 300% for France and the UK. Taxes for the US (which vary by state) are typically half as high as the lowest EU taxes (for Portugal and Greece). Note that fig. 12 is ordered in increasing rates of diesel tax rate, where the average *rate* is high about 190% (but again over 300% in the UK). Road fuel taxes contribute the overwhelming proportion of energy taxes, and raise the greatest conceptual issues, as a considerable part of these taxes are more properly considered as road user charges.

Describing and quantifying levels of fuel taxation in transition countries is more difficult, at least for non-accession countries and for all countries before 1990. Commodity taxation was primarily designed to adjust for price differences between COMECON trading partners, and to extract rents or provide transfers, rather than guide resource allocation. It is difficult collecting prices relative to the efficient level for most sectors, but as a guide, residential electricity prices were only half the long-run marginal cost (LRMC) for transition countries in 2000 (EBRD, 2001, p96). Similarly, heat prices are significantly below the LRMC in most transition countries, and the relative pattern of industrial to domestic prices is highly distorted. The second problem which continues to affect most transition countries is that the proportion of energy bills actually paid in cash is frequently low – in 2000 only 15% in Azerbaijan, 25% in Uzbekistan, 35% in Georgia, 45% in Romania, although it had risen to 85% in Russia from levels as low as 20% earlier (EBRD, 2001, p96). Commercial losses (non-billed consumption) are also high. If consumers do not have to pay the quoted price, they are effectively further subsidised.

The special case of coal

Coal is nominally untaxed in the EU (and most other OECD countries) except in Denmark and Finland, neither of which mine coal. Coal production has until recently been heavily subsidised in most significant European coal producing countries, and until recently the protection was provided by a combination of hydrocarbon taxes and above world-market domestic prices. In the early 1990s, Germany had the largest indigenous coal industry in the EU, and one of the most protected in Europe, as measured by the producer subsidy equivalent (PSE) per tonne (IEA, 1993, p38 estimates Germany's PSE as $105/tonne coal produced in 1992). Germany also paid the highest prices for coal for generation, and had the highest industrial electricity price. The UK had the lowest PSE/tonne of the European coal producers ($18/tonne of coal in 1992) but one of the highest coal prices for electricity generation. Interestingly, it also had one of the lowest industrial electricity prices of coal-intensive countries, as British coal was protected by high contract prices with the generators that were passed on primarily to non-industrial customers. Spain had an even more protected coal industry. Newbery (1995) estimated that the PSE raised the effective domestic price for coal producers about 450% above import parity in Spain (compared to the IEA's estimate of 100%), about 250% in Germany, and about 50% in the UK.

Since then, the system of supporting coal producer prices in Germany has changed so that industrial consumers (mainly power stations) can buy at import prices. Coal-backed contracts have ended in the UK, so many of the past distortions have disappeared. On the other hand, the Climate Change Levy in Britain has been carefully designed not to be a carbon tax, but an energy tax, and electricity is taxed on production, not inputs, to protect coal. Coal escapes carbon taxes (except in Denmark). Clearly coal is still treated rather leniently compared to most other fuels.

Explaining current fuel taxes

Looking at the considerable dispersion of tax rates for the same fuel in similar countries, and across similar fuels in the same country, it is hard to accept that fuel taxes have been set to internalise external costs, or to improve the efficiency of energy use. The simplest explanation is that gasoline and road diesel are mainly taxed as a means of charging for road use, as discussed below. That in turn often makes high taxes on kerosene necessary to prevent diversion to road use. Heavy oil is taxed to protect indigenous coal in some countries (e.g. the UK), and coal production (though not normally consumption) is frequently subsidised to protect coal-mining jobs. In cold climates the concept of “fuel poverty” influences domestic fuel taxation, where in

Britain fuel poverty is defined as spending more than 10% of income on energy (a problem that in the mid 1990s affected perhaps 20% of households). Thus Britain levies a lower VAT rate on domestic fuel use (5% instead of the standard 17.5%). In warmer southern climes, it is politically easier to tax domestic gas and electricity.

Finally, and especially in countries with poor tax compliance, fuel taxes are cheap and easy to collect and often attractive on those grounds. Where municipal authorities receive the income from domestic electricity and gas sales, prices are often raised (and the fuels therefore implicitly taxed) to pay for municipal services (as in Germany). Where competition and/or falling fuel prices have driven down wholesale electricity prices, some countries (Germany, the Netherlands) have imposed additional taxes to collect these rents (often in the guise of eco-taxes, to encourage a switch to green or renewable electricity). In other countries, notably the UK, energy taxes are politically so salient that they have been reduced while liberalisation drove down wholesale prices.[20]

Nevertheless, EU countries are increasingly adapting energy taxes to address environmental concerns and defending them on those grounds. Several countries have introduced comprehensive "green" tax reforms, in which environmental taxes have been raised and the revenue normally used to reduce other taxes or support renewable energy. IEA (2002a) gives more details, but briefly, Finland and Denmark introduced "carbon" taxes in 1990 and 1992, though with extensive rebates for industry and power. Norway effectively also introduced a "carbon" tax in 1991, although confined to mineral oil and again with some rebates. Italy planned much the same in 1998, but oil price increases have delayed their introduction. Sweden rebalanced its energy taxes in 1991 to concentrate the taxes on carbon and sulphur. Belgium increased energy taxes on private consumption, and Austria has imposed energy taxes on gas and electricity since 1996. The Netherlands introduced a general fuel tax and specific eco-taxes from 1995 onwards. France's attempts at restructuring environmental taxes was eventually ruled unconstitutional, while the Swiss rejected two proposals for green taxes in a referendum in 2000. Britain's Climate Change Levy has been noted. As IEA (2002a, p. xxvi) notes: 'even if these levies are often labelled "CO_2 taxes", the tax rates facing different polluters hardly reflect the carbon content of the fuels they are using."

Designing an efficient set of fuel taxes

Energy taxes are primarily input taxes, and as such fall on production as well as consumption. Standard tax theory (Diamond and Mirrlees, 1971) argues that distortions should be confined to final consumption, leaving production undistorted. In the absence of externalities or other market failures, that suggests that all indirect taxes should be value added taxes. However, externalities are prevalent, and there are good reasons for reflecting their social costs in corrective taxes. The simplest cases are where the damage done is proportional to the pollutant in the fuel. Carbon is the best example, with sulphur raising minor additional problems.

Countries that have signed the Kyoto Protocol have a choice between two equivalent efficient policies, either a carbon tax at the same rate on all fuels proportional to carbon

[20] The Fossil Fuel Levy, originally set at 10% of the pre-levy final price, is an interesting example of a tax designed to collect revenue to finance nuclear decommissioning. Because it was hypothecated to that purpose, it was not called a tax and hence not subject to the normal political bargaining.

content, or the requirement to buy carbon permits to cover the carbon content of all fuel purchases (or use). The problem in setting the tax is that there are wide differences between various estimates of the appropriate level. The original EU energy tax was $10/barrel, of which half was to be on the carbon content of the fuel, the rest being on the energy content. If it were all allocated to carbon it would amount to about US$75/tonne carbon (tC). Maddison et al (1995) estimated the shadow prices of controlling the last unit of carbon dioxide released assuming optimal abatement, where the marginal cost is $(1993) 5.9/ tC. This is only slightly less than the cost assuming 'Business as Usual', calculated as $6.1/tC. ECMT (1998, p. 70) cites estimates ranging from $2-$10/tC, considerably below the EU's original proposed carbon tax discussed above. Tol et al. (2000) reviews various estimates and argues for marginal damage costs below $50/tC. The UK Department of the Environment, Transport and the Regions decided in early 2001 to take as their working assumption a central estimate of $80/tC, with a range from $40-$160. It is worth noting that a carbon tax of $80/tC would amount to $75/tonne of coal, or 215% of the EU import price of $35/tonne in 2000. The Danish carbon tax introduced in 1992 was at a rate of 100DKK/tonne CO_2 or $38/tC, while Finland levied a carbon tax of all energy at about 500 FMK/tC or $70/tC, roughly twice as high.

Parry and Small (2001) review the literature and select a central figure of $25/tC, with range $0.7-100/tC. Even their modest range of citations suggests a range of almost 100:1, with preferred estimates differing perhaps by 10:1. Emissions trading of carbon would allow a better estimate of the marginal cost of abatement, if not the marginal damage done by emissions. A tax of $25/tC would raise the cost of gas-fired electricity generation in new CCGT plant by $2.5/MWh, or 10% of the average total cost, but by $6.5/MWh for coal-fired generation, or by 35% or more of its avoidable cost. Such a tax would probably not be sufficient to make new nuclear power competitive against gas-fired generation at current gas prices, nor would it be likely to make most renewables competitive, though it would advance the date at which they are likely to be competitive. The same carbon tax applied to transport fuel would amount to less than 2 Euro cents/litre, small in comparison to the taxes shown in Fig. 12.

Sulphur taxes and sulphur trading

Sulphur in fuel produces SO_2 but unlike CO_2 it can be largely removed by scrubbing or by flue gas desulphurisation. In addition, the sulphur content of otherwise equivalent fuels varies, and sulphur can be removed from some fuels at the production stage (in oil refineries, for example). The damage done also depends on the location and height of release and the wind direction and strength at the time. These complicate confronting users with the correct marginal social cost of damage, and relatively cruder methods are needed. Under the various international agreements, each country is now limited in the amount it can release, and the logical policy to meet this limit is one of "cap and trade".

Under this policy, the government issues or auctions permits up to the cap, that is, the total amount allowed. Some part of his total may be allocated to existing polluters as grandfathered entitlements. Trading then determines the market-clearing price, and allocates them efficiently (provided, as seems to be the case, that transaction costs and market power are low). Energy users can then choose whether to buy low-sulphur fuel, install clean-up technologies, or buy permits. The evidence (Ellerman, 2002) suggests that the resulting costs of meeting the cap are lower than anticipated, and that the cost of abatement technology falls as it becomes subject to market forces, and is no longer mandated (which before gave the suppliers substantial monopoly power).

The main objection to this approach is that there is no guarantee that the level of the cap is justified on a social cost-benefit test. Newbery (1990) noted that the limits were set to avoid exceeding environmentally determined critical loads, rather than addressing the larger damage of the effect on health. If the environmental damage could be given a monetary value, then average damage costs per tonne SO_2 could be estimated for each country, and the permits replaced by a tax on sulphur content of the fuel burned (with rebates for subsequent abatement, easy to measure by the volume of sulphur removed from the combustion gases). Denmark offers the choice of taxing the sulphur content of fuel (at 2.7 Euro/kg) with rebates for sulphur not released as SO_2 (e.g. bound in the ash), or charging emissions at 1.35 Euro/kg SO_2. Norway and Sweden also impose sulphur taxes (or taxes on sulphur-containing fuels in proportion to their sulphur content) at rates between 2.5-4 Euro/kg.

Particulates

Unburned carbon particles, or particulates, measured by PM_{10}, are the most damaging and socially costly combustion products, though SO_2 and NO_x also give rise to particulates with similar damaging effects. Large Combustion Plants are subject to emissions standards, and could be taxed on their emissions, as in Denmark. Domestic emissions are now primarily controlled by prohibiting the more polluting fuels in certain areas (smokeless zones). The main problem is dealing with transport emissions, as their damage depends on when and where they are released. In most developed countries tailpipe emission standards on new vehicles have resulted in a dramatic reduction in total levels of road transport emissions, despite the continuing increase in traffic. The cost of meeting these standards means the "the polluters pay", but not necessarily in proportion to the damage done, as increasingly tighter standards are only applied to new vehicles. This can have the perverse effect of discouraging users from replacing older more polluting vehicles by newer but now more expensive because cleaner versions. Britain addresses this problem for trucks by charging a lower annual licence fee to newer and cleaner vehicles. A sensible tax regime for this form of pollutant might therefore be to levy a tax equal to the average damage of older and more inefficient vehicles (or other sources), and give rebates on the annual licence fee for improved performance. Fines for excessive emissions can be used to more accurately target the emission charge on the small number of gross polluters.

The next question is to determine the correct level for the particulate tax. Newbery (1998) argues for estimating the social costs of the health effects of pollution by estimating the number of quality adjusted life years (QALYs) lost through premature mortality and morbidity. These costs should then be compared with what it costs the taxpayer to enable the National Health Service (in the UK or its counterpart in other countries) to achieve an extra year of quality life. The numbers used in the evaluation of transport should be consistent with numbers used elsewhere in health economics. This would enable the money raised in green taxes (which are mainly the costs of health damage) to be allocated to the National Health Service, which should be able to compensate for the quality life years lost through pollution by an equal saving of quality life years gained from improved health services.

Recent work presented in UN/ECE (2001) suggests an encouraging convergence in estimates of the mortality effects of the more damaging pollutants. Severe urban pollution reduces life expectancy, and a *permanent* increase in air pollution of 10 $\mu g/m^3$ of PM_{10} is estimated to raise the daily mortality rate by 1%. That in turn would reduce average life expectancy in Britain by 34 days (weighted by the British age distribution and based on current age-specific mortality rates). In order to relate the loss of QALYs to the annual consumption of fuel, the correct calculation is the total loss of QALYs for a one-year increase in emissions, leaving future mortality rates at the zero emission level. Newbery (2003) shows that road transport may be responsible for 4.4 $\mu g/m^3$ of PM_{10} in Britain, reducing the loss of life-expectancy per person exposed to 0.21 days per year of exposure. If we err on the high side and suppose that QALYs do not decrease with age (as they do), and take the exposed population as all 58 million people, the total number of QALYs lost by one year's traffic particulate emissions is 34,000.[21]

The UK Department for Transport assumes a value of a statistical life saved (VoSLS) in traffic accidents as £1.44 million. The weighted average age of a traffic accident if all are equally exposed is 38, and life expectancy is then 40 years. We can therefore take a statistical life as 40 QALYs, making the value of a QALY as £36,000. The UK National Institute of Clinical Excellence was reported (*Times*, 10 Aug, 2001) as tentatively accepting a figure of £30,000 per QALY, suggesting a convergence on the valuation side. At £36,000/QALY, the cost of traffic pollution is £1.2 billion/yr, negligible compared with road taxation of £27.5 billion in 2000/01 (excluding all VAT). Most (89% in Britain) of this cost is attributable to diesel vehicles, and would amount to 5.6p/litre of diesel (100 Euro/tonne, or 0.9p/km), and 0.45p/litre of petrol (10 Euro/tonne or 0.04p/km). Note that for pre-1993 vehicles the particulate tax should be 40-50% higher than this average value, and for post-1997 vehicles 50-80% lower (with diesel cars experiencing the greater improvement). These figures do not include the cost of morbidity (which is likely to be a modest fraction of mortality costs).

Nitrogen oxides

Similar principles can be used to determine standards for and taxes on NO_x emissions, as these share many of the same attributes as PM_{10} and SO_2. The US has a regional cap and trade system for NO_x which contributed to dramatically increased wholesale electricity prices in California in the summer and autumn of 2000. There the price of tradable NO_x permits rose to unprecedented levels as the annual quota became inadequate, with permits trading at \$80,000/ton at their peak, compared with \$400/ton on the East Coast (Laurie, 2001). If this approach is to be adopted the permit price needs itself to be capped at a sensible estimate of the marginal social damage done in adverse conditions (e.g. summer temperature inversions). While the US SO_2 program appears to have been a considerable success, the NO_x program clearly needs modification, but does not undermine the general claim that market-based systems have the potential to lower compliance costs considerably compared with command and control solutions.

Taxing transport fuels

Transport fuel prices in most EU countries are set at high levels as road user charges, not to

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[21] Compare this with the estimate in BeTa, the Benefits Table data base listed on the EC DG Environment web site, which assumes, without evidence, that the number of life years lost to the chronic effects of particles on mortality is 5 years.

reflect environmental externalities. Newbery (1990b, 1998) sets out the principles for designing a set of road user charges, and the likely levels of fuel tax required in the absence of road pricing. Newbery (2003) estimated that an appropriate level for the UK in 2000 would be 60 Euro cents/litre for gasoline and 67 Euro cents/litre for diesel. Fig. 12 shows that in the EU in 2001, only Italy, Denmark, Sweden, Finland, France, Netherlands and Germany were levying gasoline taxes above 50 Euro cents/litre while the UK was charging some 25% higher than justified. The only OECD country charging the target level for diesel or above was the UK, with Norway the next with a shortfall of about 10%.

Concluding remarks

Energy prices have a considerable effect on energy consumption, but targeted taxes on environmental pollutants are considerably more effective at encouraging sustainable energy use. The wide range of energy intensities, and the even wider range of emissions per unit of activity (GDP, vehicle km, kWh) suggest that suitable taxes and standards can have powerful effects on environmental emissions. The transition countries have dramatically reduced energy use and emissions, though in most cases as a result of the collapse of economic activity rather than improved efficiency. Nevertheless, as energy prices are gradually adjusted to world market levels, and new environmental standards accepted (often as part of accession agreements), incentives for efficiency improvements are being introduced.

Liberalisation and privatisation make market instruments (permit trading and eco-taxes) increasingly preferable to command and control solutions, and in turn create a constituency for measuring the damage of pollutants and relating taxes to damage. International agreements for the transboundary acid rain pollutants have been surprisingly successful, given their very asymmetric cost distribution and the slender economic basis for the agreed levels. Combined with cap and trade markets, the marginal costs of abatement in each country should be come clearer, and with it perhaps a move to trade across boundaries to equalise these prices, potentially offering large cost reductions. Full carbon trading might on some estimates reduce compliance costs by a factor of eight.

Transport as a sector has the fastest growing source of CO_2 emissions, and also attracts the highest rates of tax. In some countries these high and growing transport fuel taxes are defended as necessary on environmental grounds, but a careful assessment casts doubt on this claim, except in North America (where such claims are in any case absent). Transport taxes in most European countries are higher than emissions taxes alone would suggest, though (with the exception of the UK) below the combination of road user charges and emissions taxes. Diesel fuel is normally and inefficiently less heavily taxed than gasoline. The energy uses that stand out as under-taxed in most countries are coal (with notable exceptions in countries that have introduced carbon taxes), and households, where distributional concerns obstruct incentives for improved energy efficiency.

Pessimists (and economists) continue to criticise the irrationality of energy tax policy, but optimists can point to the steady improvement in our understanding of the costs and benefits of reducing pollution, and the resulting improvement in the design of emissions policy.

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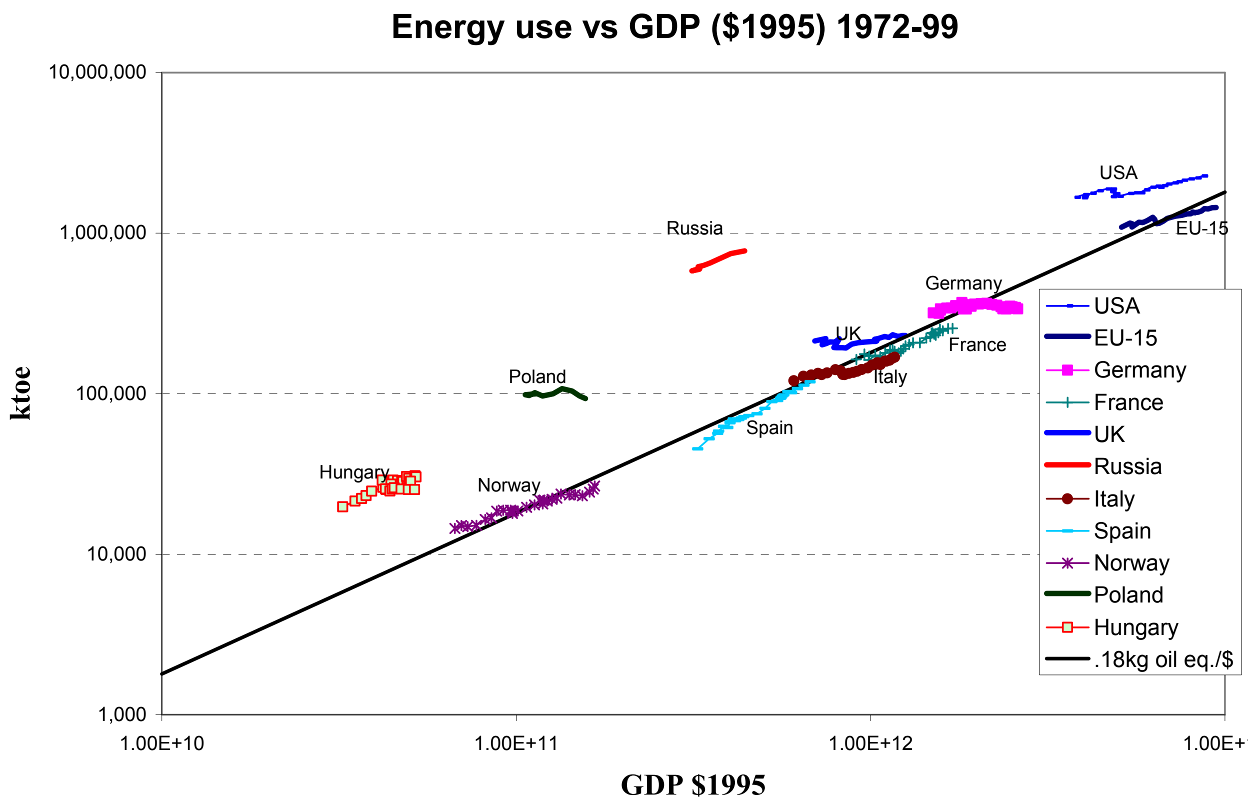

Energy use vs GDP ($1995) 1972-99
ktoe
10,000,000
1,000,000
100,000
10,000
1,000
1.00E+10
1.00E+11
1.00E+12
1.00E+13
GDP $1995
USA
EU-15
Russia
Germany
UK
France
Italy
Poland
Spain
Hungary
Norway
USA
EU-15
Germany
France
UK
Russia
Italy
Spain
Norway
Poland
Hungary
.18kg oil eq./$

Fig 1

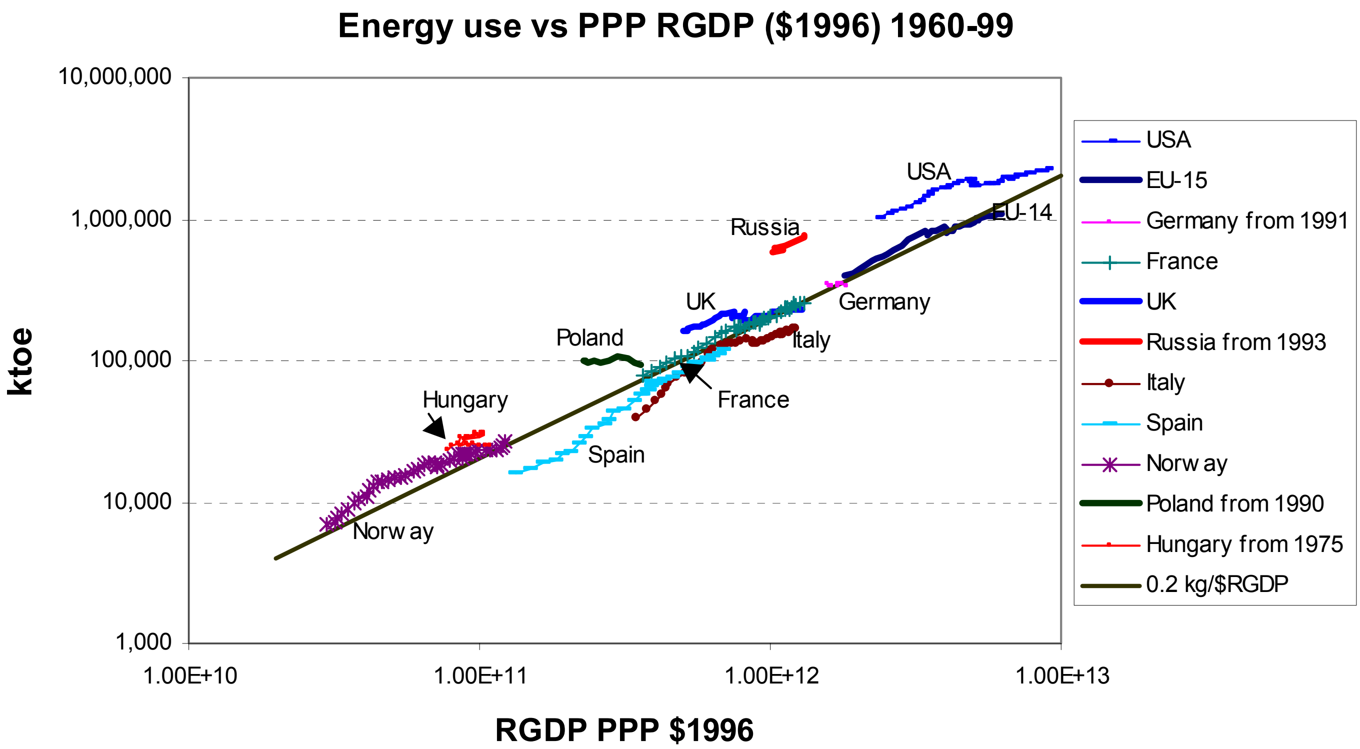
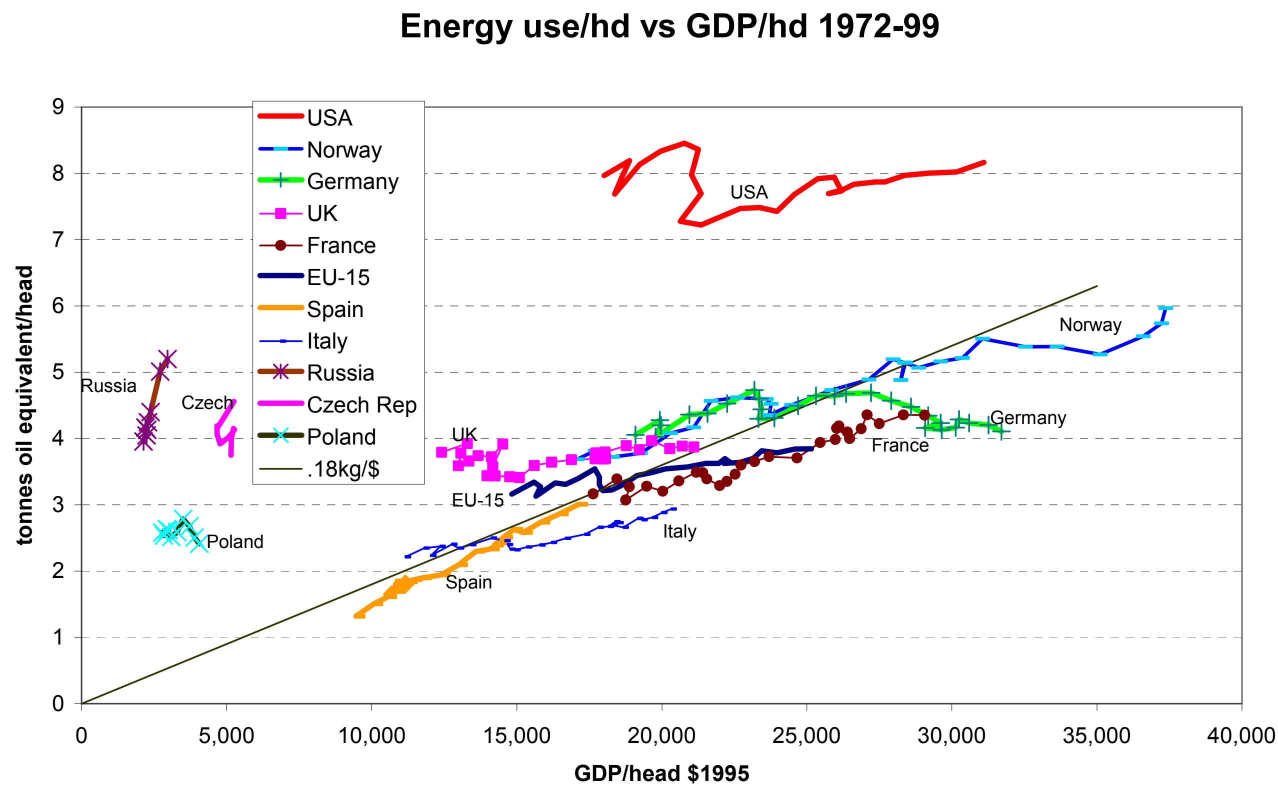
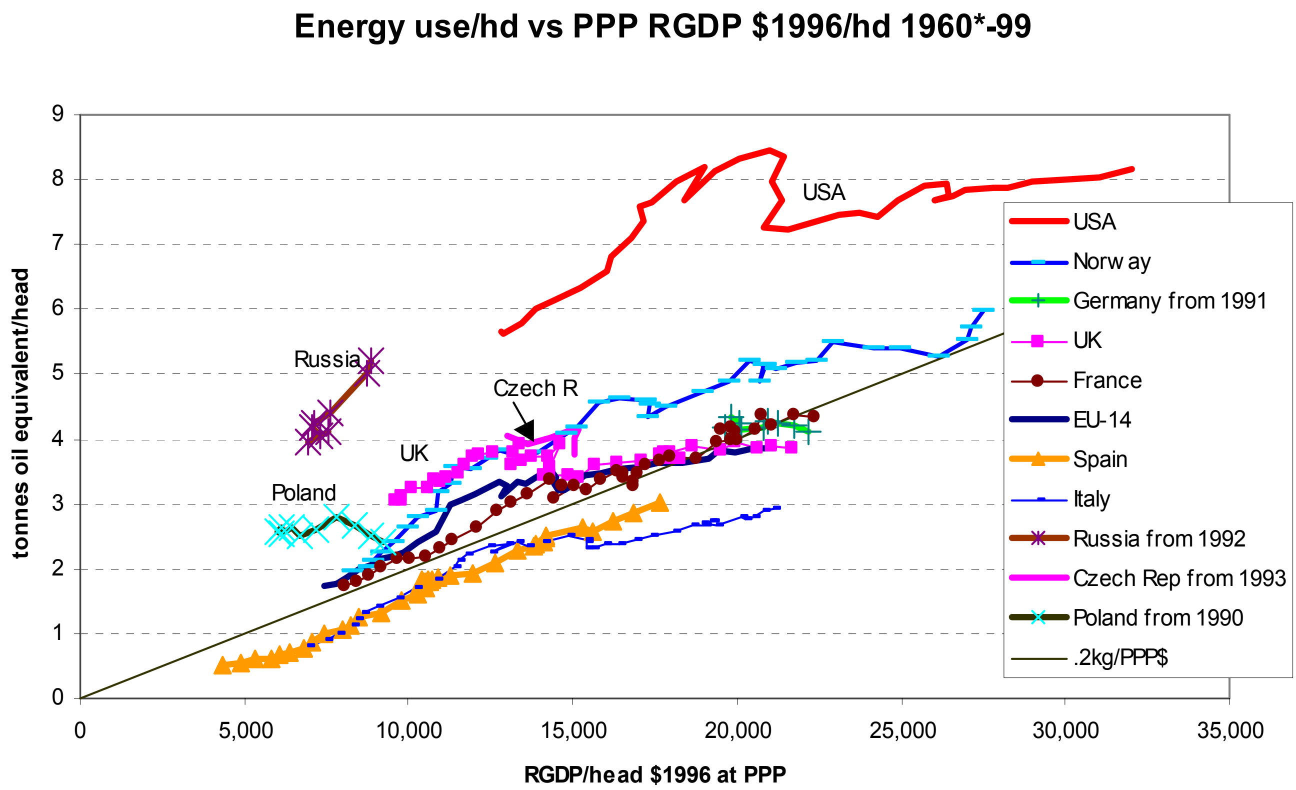

Energy use vs PPP RGDP ($1996) 1960-99
ktoe
10,000,000
1,000,000
100,000
10,000
1,000
1.00E+10
1.00E+11
1.00E+12
1.00E+13
RGDP PPP $1996
USA
EU-14
Russia
UK
Germany
Poland
Italy
Hungary
France
Spain
Norw ay
USA
EU-15
Germany from 1991
France
UK
Russia from 1993
Italy
Spain
Norw ay
Poland from 1990
Hungary from 1975
0.2 kg/$RGDP

Fig. 2



Energy use/hd vs GDP/hd 1972-99
USA
Norway
Germany
UK
France
EU-15
Spain
Italy
Russia
Czech Rep
Poland
.18kg/$
tonnes oil equivalent/head
GDP/head $1995
Russia
Czech
Poland
UK
EU-15
Spain
Italy
USA
France
Germany
Norway
0
1
2
3
4
5
6
7
8
9
0
5,000
10,000
15,000
20,000
25,000
30,000
35,000
40,000

Fig 3



Energy use/hd vs PPP RGDP $1996/hd 1960*-99
USA
Norw ay
Germany from 1991
UK
France
EU-14
Spain
Italy
Russia from 1992
Czech Rep from 1993
Poland from 1990
.2kg/PPP$
tonnes oil equivalent/head
RGDP/head $1996 at PPP
Russia
Czech R
UK
Poland
USA
0
1
2
3
4
5
6
7
8
9
0
5,000
10,000
15,000
20,000
25,000
30,000
35,000

Fig. 4

Cross-section relation between average energy intensity and average energy price 1993-99

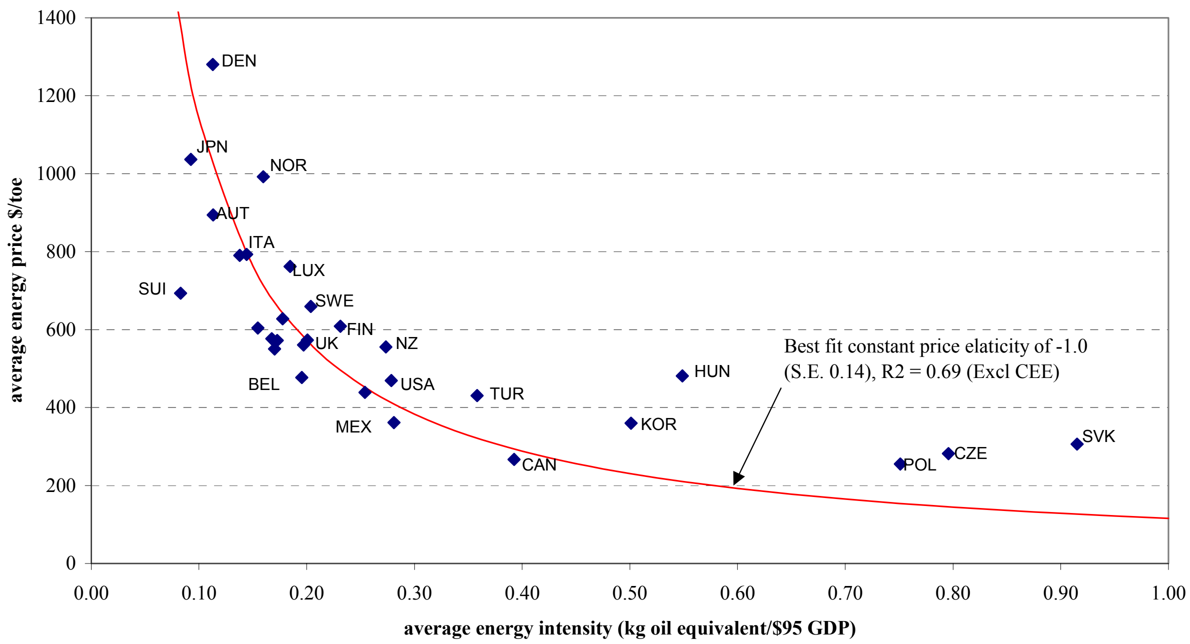


Fig. 5

CO2/tonne oil equivalent

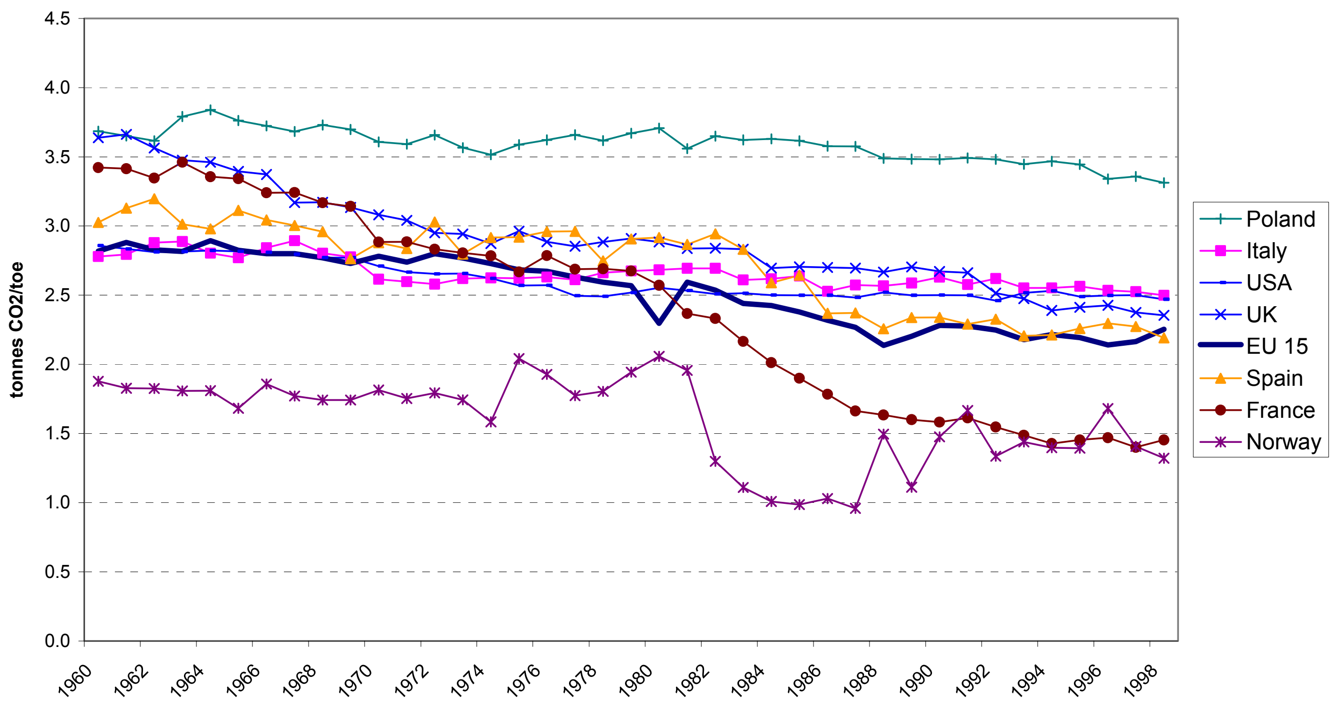


Fig. 6

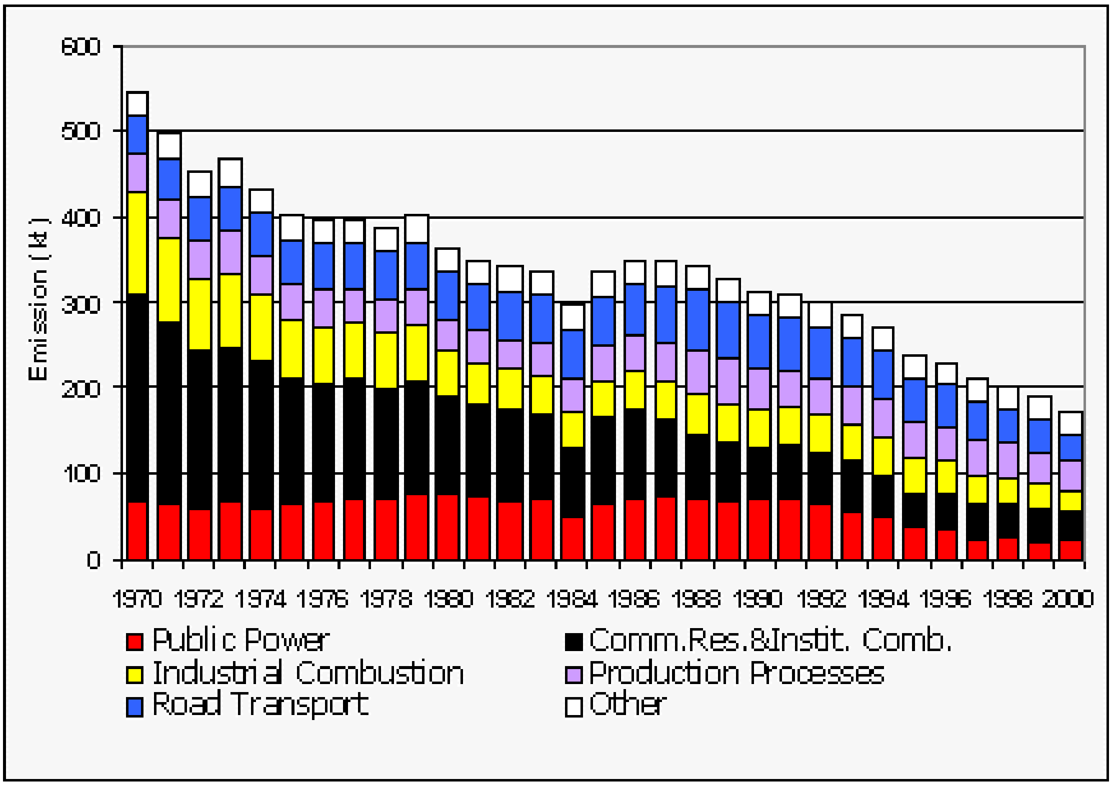


Fig. 7 7 UK emissions of particulates

Source: National Atmospheric Emissions Inventory http://www.naei.org.uk/

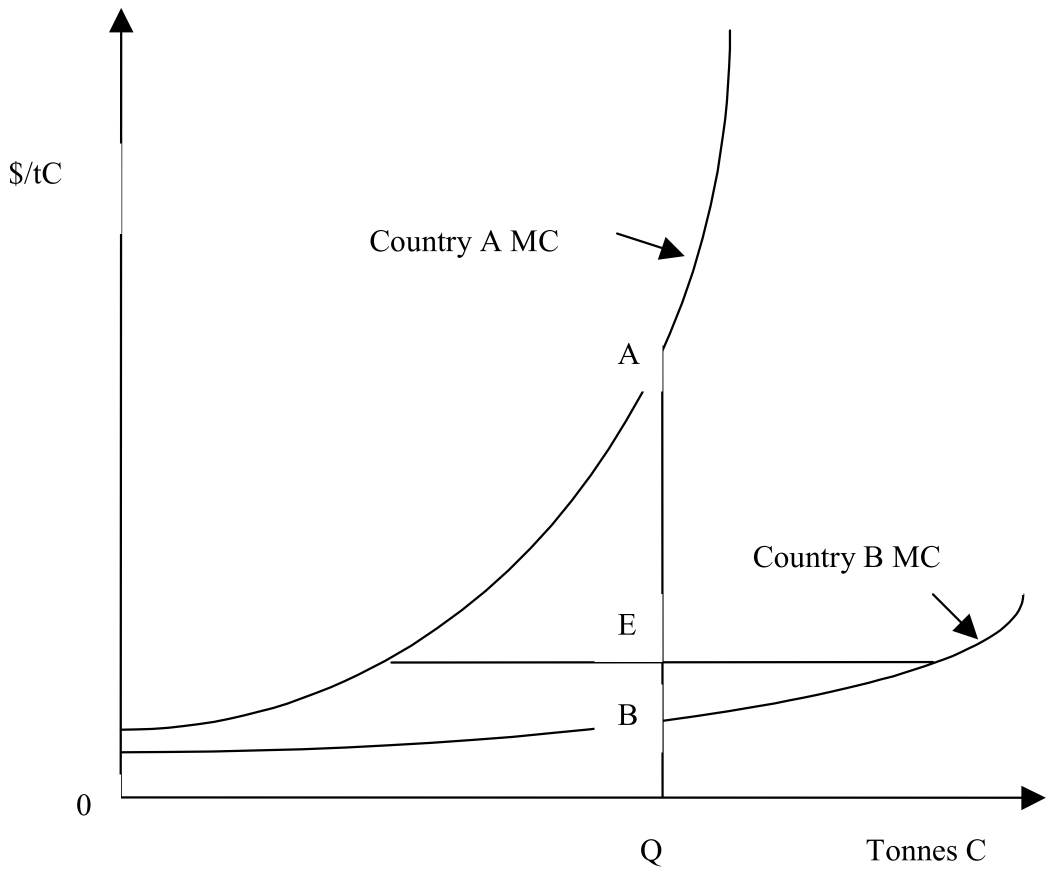
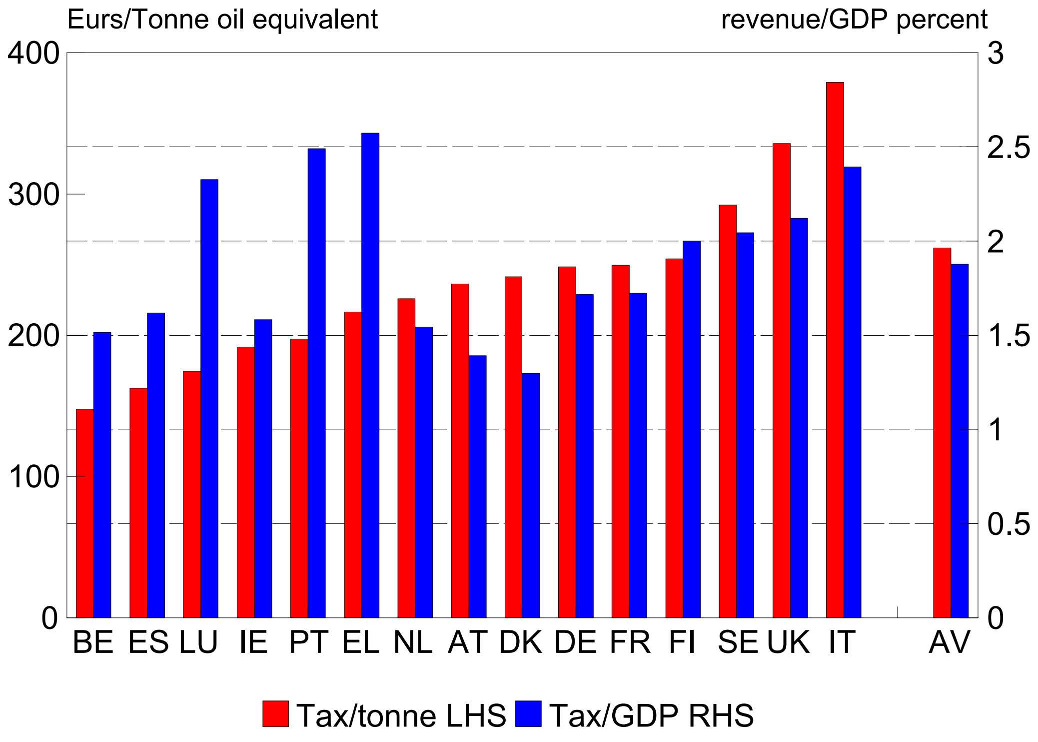


Fig. 8 The benefits of emissions trading

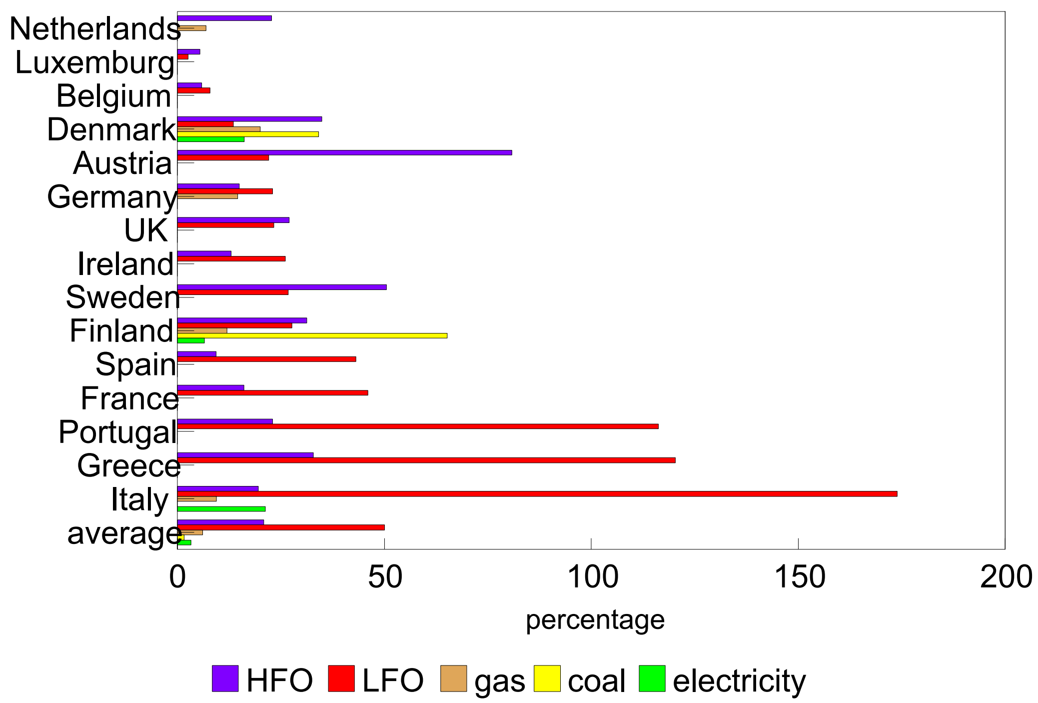
Average mineral oil excise 1997



EC Excise Tax Duty Tables, July 2001
ranked by tax rate

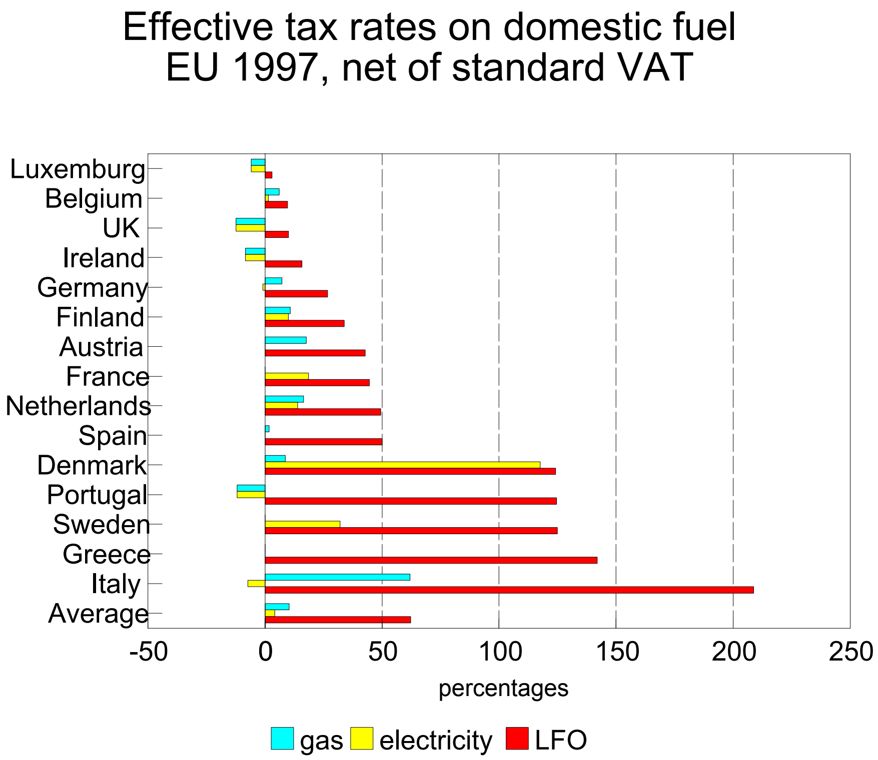
Fig. 9

Tax rates on industrial fuels
EU 1997, excluding VAT



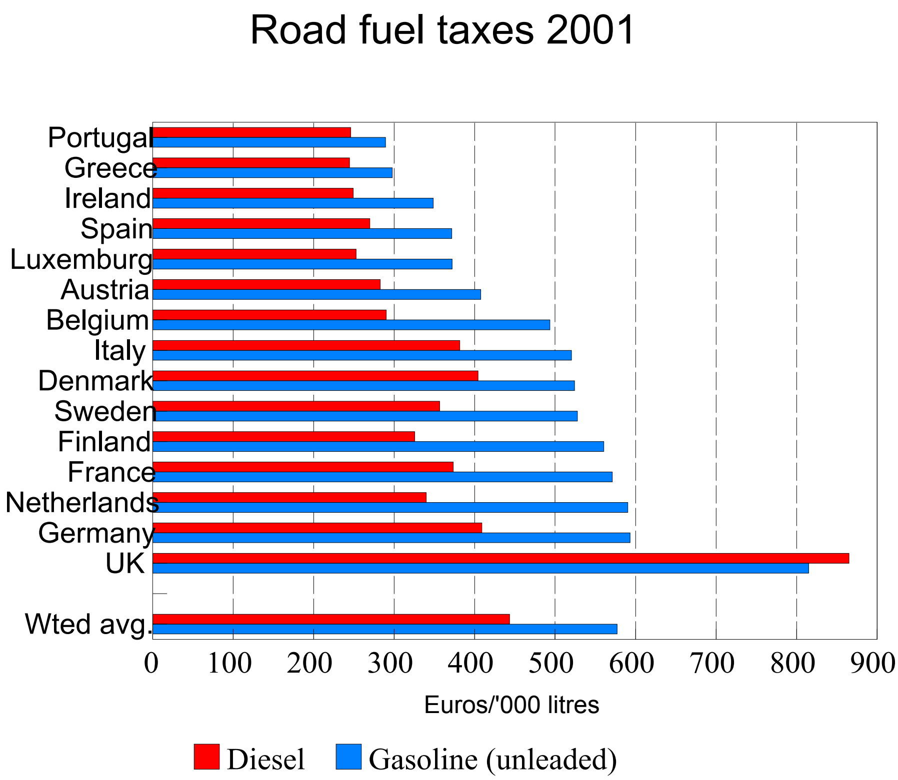
IEA Energy Prices and Taxes
ranked by tax on LFO

Fig. 10



IEA Energy Prices and Taxes

Fig. 11



European Commission

Fig. 12

Table 1 Energy intensity relative to average EU energy intensity, 1972-99

market exchange rates, $1995

Country	1972	1980	1990	1999	change 72-99
EU-15	1.19	1.07	0.92	0.85	-28%
Austria	0.84	0.75	0.66	0.61	-27%
Belgium	1.48	1.23	1.05	1.08	-27%
Denmark	0.90	0.79	0.61	0.56	-38%
Finland	1.44	1.44	1.20	1.19	-18%
France	1.00	0.91	0.85	0.83	-17%
Germany	1.18	1.10	0.87	0.72	-39%
Greece	0.76	0.85	1.10	1.13	49%
Ireland	1.52	1.28	1.10	0.82	-46%
Italy	1.11	0.94	0.82	0.80	-27%
Luxembourg	3.06	2.24	1.42	0.85	-72%
Netherlands	1.35	1.20	0.99	0.87	-36%
Portugal	0.72	0.80	0.93	1.06	48%
Spain	0.78	0.93	0.92	0.98	26%
Sweden	1.30	1.18	1.11	1.06	-18%
UK	1.70	1.40	1.14	1.02	-40%
Norway	1.20	1.08	0.98	0.89	-26%
Bulgaria		13.56	10.73	8.74	
Czech Republic			4.83	4.05	
Hungary	3.39	3.59	3.15	2.72	-20%
Poland			4.88	3.31	
Romania		9.21	8.80	6.30	
Slovak Rep			5.65	4.56	
Russia				10.18	
United States	2.46	2.11	1.64	1.46	-41%

Source: World Bank Development Indicators
Note: annual figures are relative to the EU-15 energy intensity averaged over the period 1972-99

Table 2 Index of Industrial energy intensity, 1971-2000

1995=100

Country	1971	1980	1990	1997	2000	change 71-00
EU-15	*179.4*	136.40	105.00	98.70	94.60	-47%
Austria	176.70	151.70	110.90	102.60	90.10	-49%
Belgium	151.79	119.72	95.43	109.63	114.75	-24%
Denmark	218.27	164.16	103.33	93.31	80.40	-63%
Finland	112.84	100.79	113.19	97.71	81.62	-28%
France	159.42	130.11	94.01	98.72	88.54	-44%
Germany	161.03	154.50	111.45	95.40	90.57	-44%
Greece	115.85	108.05	99.68	107.56	100.47	-13%
Ireland		275.45	145.50	82.10	54.60	
Italy	179.47	132.50	108.13	99.83	98.19	-45%
Luxembourg	265.35	230.72	135.62	89.65	77.55	-71%
Netherlands	118.08	128.75	113.73	96.45	96.49	-18%
Portugal	111.71	102.45	105.95	100.07	108.70	-3%
Spain	128.51	112.45	93.78	99.91	106.34	-17%
Sweden	147.64	131.10	106.05	95.60	82.22	-44%
UK	209.69	143.45	105.42	100.67	99.70	-52%
Norway	249.76	202.41	125.02	89.69	102.51	-59%
Bulgaria						
Czech Republic			84.10	77.57	64.17	
Hungary		179.83	144.31	87.28	57.51	
Poland			124.12	85.83	58.39	
Romania						
Slovak Rep			164.31	88.90	93.33	
Russia						
United States	220.65	176.48	112.05	90.28	80.11	-64%

Source OECD *Energy Balances 2000*
Note: The figure for EU-15 for 1971 is an estimated weighted average of the 15 individual countries