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SUBSIDIES AND THE ENVIRONMENT: THE STATE OF KNOWLEDGE

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Introduction

This study surveys the literature on subsidies and their impact on the environment in five economic sectors — agriculture, fisheries, forestry, energy and transport[1] — as well as on irrigation water. It aims to answer the following questions:

- what are the different ways that subsidies in each sector are defined and measured?
- what country-by-country data on subsidies to producers or consumers are either already available to researchers or could be used to construct such estimates?
- what methodologies are in use or are available for estimating the impact of subsidy removal on the environment in each sector and what results have they produced, if any? and
- what significant gaps or problems exist in the data on subsidies in each sector and what additional research is needed to establish an adequate database on subsidies and to measure the environmental benefits of subsidy removal?

The study was guided by terms of reference that called for consideration of six different categories of subsidies: budgetary transfers, market price support, subsidised and concessional credit, under-priced materials, water and energy, foregone tax revenues, and foregone resource rents plus uninternalised externalities. It represents an inventory of existing conceptual frameworks, data sources and methodologies available for both documenting subsidies and for determining the impact of subsidy reform on the environment. The category of foregone resource rents is relevant to the definition of subsidy in some of the economic sectors covered, but not in others. The study notes the sectors in which each of these categories plays a role in the definition of a subsidy.

Each of these sectors has distinct structural characteristics that have influenced the methods used to define and measure subsidies as well as the nature of the research done to analyse the environmental benefits of subsidy removal. No less than six different methods have been used to define and measure subsidies in the six sectors, and more than one method has been used in every sector except in the fisheries sector, for which the aggregation of various support programs is accepted as the only method for quantifying subsidies. Aggregating the relevant support programs is at least one method for determining the size of the subsidy in agriculture, fisheries, forests and energy sectors (although analysts may differ in some cases over what specific programs should be included).

The discussion of data and analytical methods in regard to each of these categories of subsidies is not related to the analysis of environmental impacts. Inclusion of a particular category of subsidies in the survey of data available on subsidies in a given sector does not imply any conclusion regarding its impact on the environment. Quantifying the impacts of subsidies — and the impact of subsidy removal — on the environment is an analytical challenge in each of these sectors. We know that subsidies create incentives for environmentally damaging activities, but it is often difficult to isolate the effects of the subsidies from other drivers, including perverse economic incentives from lack of property rights. OECD (1998a) suggested in its analysis of studies on agriculture, energy and transport subsidies that "orders of magnitude of the effects of support measures on the environment are often all that can be determined."

This survey shows that some methodologies for estimating the environmental impacts of subsidies or subsidy removal show promise in refining such estimates, even if significant uncertainties remain. Applied to existing data or to data that could be collected with some additional effort, these methodologies should be able to provide more precise and convincing estimates than now exist in the agriculture, fisheries, and energy sectors. For transport infrastructure and irrigation water the picture is less clear.

Agriculture

Distinguishing characteristics of the sector

- By and large, the government does not provide either the primary capital investment or the crucial natural resources used in agriculture, as in the cases of forests, fisheries and transport. Irrigated agriculture represents an obvious exception to this generality as noted in the discussion of that sector below.

- For the same reason, resource rents are not an issue in defining and measuring agricultural subsidies.

Defining and estimating agricultural subsidies

A number of measures of agricultural subsidies have been developed for various purposes. The measures most often used, however, are the Producer Support Estimate (PSE), the Consumer Support Estimate (CSE) and the Aggregate Measure of Support (AMS).[2] A full explanation of the concepts, methodology, interpretation and guidelines for the use of these OECD support indicators can be found in *Methodology for the Measurement of support and use in policy evaluation.*

Statistical measures of agricultural subsidies vary in purpose, depending on whether they are intended to measure subsidies from all sources or only those subsidies that fall under WTO disciplines. The measure often used to estimate total support for agriculture is the PSE, which takes into account all transfers to agricultural producers from all sources, including both budgetary support (government services, direct payments and export subsidies) and market price support (Cahill and Legg, 1989-90; OECD, 2001). Market price support measures the benefit to producers of the price gap creates between domestic prices and world prices for each commodity created by both trade and domestic policies. The total PSE is aggregated from the PSE for specific commodities and is expressed in three different ways: as the total value of transfers to the commodity produced; as the total value of transfers per unit of commodity produced, and as a percentage of the total value of production, including the transfers.

The PSE as percentage of total value is the only one that permits comparison across economies of very different size, so it is useful in providing a measure of the degree of distortion produced by subsidies in the agricultural sector in each country. In the 1998-2000 period, the average PSE in OECD countries was 37%, and ranged from a low of 1% in New Zealand to 71% in Switzerland (OECD, 2001). Thus, the PSE does not measure trade distortion *per se* but allows comparisons of gross transfers across countries (Cahill and Legg, 1989-90; OECD, 2001). The aggregate PSE for all commodities expressed in absolute terms is a useful measure for documenting trends in agricultural subsidies at the national and international levels.

Another measure of agricultural subsidies is the Consumer Support Estimate (CSE), which estimates the value of these gross transfers from or to domestic consumers of agricultural products. When the CSE is negative, especially as a result of price supports to farmers, it represents a tax on consumers. A CSE can similarly be expressed as an absolute figure for each commodity, as a per unit of production figure, or as a percentage of value figure. The percentage of value expression of both PSEs and CSEs are influenced by world markets through their market price support components: when world prices for agricultural commodities rise, percentage PSEs and CSEs fall (Cahill and Legg, 1989-90). These subsidy estimates fluctuate constantly, regardless of government agricultural subsidy policies and therefore do not necessarily reflect the trend in government policies.

The measure of agricultural subsidies in the negotiations for the Uruguay Round Agreement on Agriculture (URAA) and in the current agricultural trade negotiations, however, is the Aggregate Measure of Support, or AMS (Hamsvoort, 1994; Nelson, 1997). It is a measure only of domestic subsidy programs, including market price support, excluding export subsidies. It also excludes programs that do not fall under WTO disciplines (programs that are regarded as only minimally trade distorting, that take farmlands out of production, or that do not exceed the *de minimus* standard of 5% or 10% of the member's total value of production, depending on whether it is an industrialized or developing country).

Thus the AMS is a measure of all domestic subsidy programs that are currently disciplined by an agricultural trade liberalisation agreement. They reflect current policy preferences rather than a theoretical or empirical judgement about the degree of trade distortion associated with any particular domestic subsidy program. One other difference between the AMS and the PSE is that the AMS uses a method of calculating domestic support that does not fluctuate with world prices (Nelson, 1997). Finally, the AMS includes the total of non-commodity-related subsidies (*i.e.* those to production inputs, such as fertilizer water, and grazing subsidies) as a separate component of the whole, rather that allocating them across all relevant crops in relation to their value (Nelson, 1997).

The total support estimate (TSE) is the cost to consumers and taxpayers of support to the agricultural sector as a whole (*i.e.* government services such as agricultural research) and not just to individual producers alone. The TSE includes the PSE, CSE and GSSE (General Services Support Estimates), which comprise the full range of state-funded services of value to the agricultural sector.

The Effective Rate of Protection (ERP) is the percentage difference between the value added per unit of output at domestic prices and value added at world prices measured in a common currency. It is a more comprehensive measure of the distortion of incentives in the agriculture sector from all policies affecting the sector. It takes into account explicit or implicit taxes on agriculture as well as border protection and other assistance to the inputs used, such as water and energy, and to the activity rather than simply to the product itself. It is far more complex to calculate and it is seldom used to compare rates of agricultural subsidisation.

The Trade Distortion Equivalent (TDE) converts transfers to the agriculture sector in countries that use supply controls into a measure of trade distortion. The TDE attempts to identify the "shadow price" which would have brought forth the actual level of production occurring under supply controls. Then it measures the price gap between the actual price for an agricultural good and this hypothetical shadow price (Cahill and Legg, 1989-90). Another trade-related measure of subsidy is the nominal assistance co-efficient (NAC) for producers, which measures the gap between domestic prices (measured at the farm gate) and world prices.

Country-by-country agricultural subsidy data available

The OECD produces estimates of support to agriculture for all OECD countries. Aggregate data for the years 1986 through the latest calendar year are available on line or on a CD-ROM. The databases include Total Support Estimates (TSE), including total PSE, CSE, and GSSE in the country, and estimates of Market Price Support (MPS), CSEs, and PSEs by commodity. PSE estimates include MPS and budgetary payments broken down by type of payment (whether based on unlimited output, limited output, area planted or animal numbers, historical entitlements, input use, input constraints and overall farming income).

In an analysis of policies regarding agriculture in "emerging and transition economies," OECD (*2001b*) estimates TSEs, PSEs, CSEs, Producer NPCs and Producer NACs for eleven former socialist economies from 1991 to 2000. The data show that five of the eleven former socialist economies had negative PSEs for at least three of the five years from 1991 through 1995, meaning that government imposed a net tax on agricultural producers. Russia and several other transitional economies had PSEs in the mid- to late-1990s that were often comparable to the lowest rates among OECD countries. Low or negative PSEs in countries that have maintained support programs that were often quite high have been explained by a combination of very high transaction costs and price data that are often unreliable (Meyers, 1996; Harley, 1996).

The WTO collects country-by-country data on agricultural export subsidies for 25 countries, as well as figures on domestic support to agriculture for 56 countries. Its latest figures cover each of the years 1995 through 1998. The totals for domestic agricultural support are broken down into five categories: “green box”, special and differential treatment, blue box, support measures within *de minimus* levels, and total AMS subject to reduction under the Uruguay Round Agreement on Agriculture (URAA). The domestic support data are highly aggregated, and are not subject to any check by the WTO Secretariat except through Trade Policy Reviews every four years or longer. It is unclear whether the WTO will continue to update the existing database on domestic support for the 56 countries annually in the coming years.

In some ways more useful than the country-by-country figures on domestic support to agriculture published by WTO are the Trade Policy Reviews of its member states, which go into considerable detail in regard to support for the agricultural sector. The four largest traders (the EU, the United States, Japan and Canada) are examined every two years; the next 16 countries are examined every four years, and the remaining countries are reviewed every six years, except for least-developed countries, which may be reviewed even less often. Each trade policy review includes a full report by the WTO Secretariat, which contains figures for various types of agricultural subsidies, such as agricultural credit subsidies, subsidised insurance services for the agricultural sector, assistance to agricultural inputs and electricity (*e.g.* Mexico).

The United States Department of Agriculture’s Economic Research Service (ERS) provides a database on diskette for PSEs and CSEs for 30 countries and the EU for the years 1962-1992. The countries covered are all developing countries and socialist countries (later “countries in transition”) except for Canada, the EU, Japan and the United States. The database is available from the US National Technical Information Service (NTIS). These data were developed for the purpose of modelling agricultural liberalisation scenarios for the Uruguay Round negotiations, and the ERS has no plans to repeat the exercise.

The ERS (Burfisher, 2001) has used data from the OECD’s 1998 PSE database and combined it with WTO export subsidy notifications and applied tariff data from the Agricultural Market Access Database to generate a new data set on domestic subsidy expenditures as a percentage of value of production (net of subsidies) for 11 OECD countries and the EU. These data are organised into five categories (fixed payment per unit of output, fixed payment per unit of intermediate input, direct and whole-farm payments, capital-based payments, and other policies deemed by the authors to have minimal trade impacts). However, they might also be useful in the comparative analysis of environmental impacts of agricultural subsidies. The managers of the GTAP Database have also aggregated agricultural export subsidies, output subsidies, input subsidies, land-based payments and capital-based payments into the database for GTAP 5 (Dimaranan and McDougal, 2002).

Separate compilations of data on the subsidisation of fertiliser and pesticides may be useful, even for countries in which most of the value of these subsidies are already captured in agricultural PSEs. Budgetary subsidies and tax rebates for fertiliser purchase are compiled by the OECD for all member countries, although fertiliser subsidies may not always be captured in the agricultural PSE data if they are the result of under-pricing of fertiliser rather than payments to farmers to cover fertiliser purchases.

The FAO once published country-by-country data on explicit subsidies for urea fertiliser (in 1995 USD per metric ton) for the period from 1971 to 1990, but then stopped collecting these data. These historical data are available for 18 countries, including six OECD countries, organised in four five-year periods (World Bank, 1997). The FAO continues, however, to track fertiliser prices and the status of subsidy programs, if any, for 77 states. This database indicates whether fertiliser subsidies are implemented through fixed prices, through payments to manufacturers, suppliers or farmers, or through more than one of these alternatives. Of the 77 countries covered by the FAO database, 39 either never had fertiliser subsidies or ended those subsidies in the 1980s or 1990s.

Additional detailed country-by-country data on fertiliser subsidies in Asia-Pacific countries are maintained by the Fertiliser Advisory, Development and Information Network for Asia and the Pacific (FADINAP), co-sponsored by ESCAP, the FAO and UNIDO and located in Bangkok, Thailand. Some earlier fertiliser subsidy data on Asia-Pacific countries are provided in a collection of papers for a regional workshop on the subject co-sponsored by ESCAP, FAO and UNIDO (FADINAP, 1996; Isherwood, 1996).

India is one the most important fertiliser-subsidising countries, with nearly USD 2.5 billion spent annually on such subsidies as of 1992 (World Bank, 1995). Its fertiliser subsidies are reported in line 2852.03.101 of the budget of the Department of Fertilisers. Indonesia’s fertiliser subsidies from 1987 to 1998, which were divided among fertiliser producers, distributors and farmers, were reported by Soedjais (1999) in rupiah and in terms of controlled prices in relation to paddy prices from 1993 through 1999. The Asian

Development Bank (2000) has collected data on Pakistan's fertiliser subsidies, which have been cut by 98% from their 1989 high point, for the years 1986 through 1996. These data cover absolute amounts of fertiliser subsidies and the percentage of the fertiliser price subsidised for both locally manufactured and imported fertiliser. Farah (1994) provided pesticide subsidy rates for several African countries.

Methodologies for measuring potential environmental impacts of subsidy removal

This section reviews the results of recent studies that have studied the relationship between support to agriculture, or to agrichemicals specifically, and certain environmental impacts. While most of the studies discussed here focussed on fertiliser and pesticide use, this variable is only one of a number of environmentally significant effects that can result from changes in support to agriculture. Other environmental issues related to agricultural subsidies include water consumption, nutrient pollution from livestock manure, soil loss, forest conversion and biodiversity loss. The effects that agricultural subsidies have on the environment have been the subject of considerable debate. It has been argued that, in the absence of adequate regulatory constraints, production and input subsidies draw more capital into agriculture by making it more profitable, resulting in more land being converted from forests and wetlands into agriculture than would be the case in an undistorted market (Runge, 1994). Some authors have also asserted that subsidies provide price signals to farmers to farm more intensively by growing the same crop year after year instead of rotating crops, causing declining soil productivity and requiring excessive use of fertiliser and pesticides (Runge, 1994; Faeth, 1995). Quantifying the impacts of subsidies or their removal, however, presents methodological problems, including separating the subsidy effect from the effects of other policy changes and having to make generalisations about impacts that vary according to agri-environmental conditions of specific sites, as well to country-specific environmental regulations (OECD, 1998a; OECD, 2000).

Nevertheless, data on fertilizer and pesticide use, and on agricultural subsidies at the national level, can be used to assess the degree of correlation between agricultural subsidies or their removal and farm management choices affecting the environment. Most of the research undertaken in this regard has focused on chemical pollution. A number of studies have observed a causal linkage between production-linked subsidies and intensification of agriculture, particularly in the context of EU agricultural policy (Heerink *et al.*, 1993; Nutzinger, 1994). Estimates of the impact of subsidy reform on pesticide use and fertiliser use or nitrogen balances can be derived from comparisons across countries over a given period of time. Such comparisons could test the correlation of PSEs with the environmental indicator against correlations between other independent variables with the same indicator. The only studies undertaken thus far have been cross-sectional studies of the aggregate correlation between average PSEs and average levels of fertiliser and pesticide use in a number of countries over a period of years. The PSEs in 11 OECD countries and the EU have been shown to be strongly correlated with the amount of nitrogen fertiliser used per square kilometre in those countries, although phosphate fertiliser is only weakly correlated with the PSEs (Norway Ministry of Agriculture, 1999). An earlier comparison of the same OECD countries and seven major non-OECD agricultural countries in terms of both PSEs and chemical fertiliser use showed a similar correlation between the two (Anderson, 1992) that was visually apparent. Another study (Harold and Runge, 1993) used linear regression of linkages from the PSE, calculated without livestock products included (because they do not use fertiliser directly) to fertiliser use in 40 countries based on data for a six year-period. They found that a one-unit increase in this modified PSE across all the countries studied would result in an increase in fertiliser use of 15.4 kg per hectare per year, whereas six other independent variables (prices for corn, wheat and rice and per capita GDP) were found to have no statistically significant bearing on fertiliser use.

The increase in direct payments to farmers in OECD countries from 5% of total support in 1990 to 25% by 1998, and the corresponding reduction in the share of market price support, has provided several investigators with an opportunity to quantify the environmental benefits of agricultural policy reform on the basis of empirical observation. The OECD (1998b) observes that when agricultural price supports are reduced, farmers respond by farming less intensively, and that the level of pollution of groundwater and surface water is reduced. In the case of Finland, the reduced profitability from applying fertilisers from the relative loss of market price support brought a reduction in use of nitrogen-based fertilisers from 92 kg per hectare in 1995 to 80 kg per hectare in 1999, and a 50% drop in phosphate fertiliser use per hectare in the same period (Steen, 2000). OECD (2001a) has also shown that the period of reduced price support subsidies in OECD was accompanied by a decline in pesticide use. Even when the correlation between subsidies and use of chemical inputs can be documented, it is very difficult to establish a quantitative relationship between subsidies and the quality of the environment. Although reducing the stresses (fertilizer and pesticide) translate into improvement in water quality and other indicators of environmental status, the latter changes generally occur over longer time spans. Furthermore, many environmental effects become evident only after a threshold of change has been reached. For example, changes in farm policy could reduce the rate at which phosphorus is introduced into a lake from surface runoff, but the eutrophication of the lake may not end until the level of phosphorus load is reduced by 40% (Brouwer, 2002).

Studies that attempt to quantify the environmental impacts of a multilateral trade liberalisation agreement on particular agricultural sub-sectors raise additional issues beyond those associated with a single country subsidy reform case study or a cross-sectional study based on subsidy reform at the national level. Trade liberalisation is not the primary factor in determining trade patterns, so assessments of trade liberalisation agreements, whether *ex ante* or *ex post*, must distinguish the effect of trade liberalisation on changes in trade from other powerful factors, such as relative price changes, changes in income and consumer tastes, and exchange rates. This can be done by examining historical trends in tariffs and trade and by partial-equilibrium models which allow the analyst to isolate the effect of trade liberalisation from other effects in a particular sector. Once that analytical problem has been solved, the increased exports and imports must be translated into impacts on production in the products selected for study in order to estimate the changes in the level of stresses on the environment. A joint study by the OECD's Directorate for Food, Agriculture and Fisheries and the Directorate for the Environment (OECD, 2000) projects trade liberalisation-induced changes in the production of wheat, coarse grains and rice in eight OECD countries, based on its Aglink model, and shows how those changes are related to rate of pesticide use per hectare and in nitrogen surplus per hectare for the same countries. It shows that the increased production from trade liberalisation will take place in those OECD countries with lower indices of agri-chemical pollution, while the EU and Japan, who have considerably higher indices of agri-chemical pollution, are projected to lose production under trade liberalisation. However, it does not try to quantify the changes in pesticide and nitrogen use likely to result from trade liberalisation.

A key issue in the *ex post* analysis of the environmental impacts of trade liberalisation agreements on specific agricultural sub-sectors is whether the additional exports that may be attributable to such an agreement were on a scale large enough to alter the level of production in a way that bears on the environment. The extremely limited character of the agricultural trade liberalisation achieved at the global level thus far is indicated by the fact that the value of agricultural commodity exports since the Uruguay Round Agricultural Agreement (URAA) have sharply declined, contrary to the general view that trade liberalisation should raise global commodity prices (Convention of Biodiversity, 2002).

A study of the impacts of the North American Free Trade Agreement and the Uruguay Round Agreement on Agriculture on North American beef, corn and tomato sub-sectors (Porter, 2003) suggests that the liberalisation in most agricultural sectors in those two agreements was too small to make a difference in production patterns. He observes that the production effects of increased exports in each of the three sectors were reduced to negligible proportions by the relatively small size of the incremental exports in relation to domestic production, major increases in yield that have halted growth in area planted, and by the relative unresponsiveness of the corn and beef sub-sectors to price signals. At least in most sub-sectors and in most countries, therefore, the modest liberalisation of agricultural trade achieved thus far is unlikely to have had any significant impacts on the environment, either positive or negative.

Economists try to capture not only the direct effects on trade patterns and production in specific agricultural sub-sectors, but also the effects on all sectors of the economy and to measure second and third-level impacts. Second-level impacts are those resulting from interactions between and among markets and production, not only in agriculture but in other sectors of the economies. The third-level impacts are the economic equilibrium, or economic adjustment effects, resulting from first- and second-level impacts, such as changes in consumer spending and employment patterns. For these purposes, analysts use computable general equilibrium (CGE) models. The CGE model is run twice, first to simulate conditions in a “base year” without the trade liberalisation agreement, and then a second time with all other macroeconomic conditions in that same year remaining the same but simulating the trade liberalisation agreement in effect (Gallagher *et al.*, 2002).

CGE models predict the effect of a given set of trade policy changes on the basis of a large and complex set of assumptions about relationships, especially price elasticities of supply and demand in specific industries, or in the case of agriculture, specific crops. These assumptions are in turn derived from economic theory, but the model must make a number of arbitrary assumptions about these relationships, which are highly uncertain. These models can only provide snapshots of relatively short-tem effects, moreover, rather than predict longer-term changes. A number of *ex ante* analyses of the trade effects of NAFTA were grossly inaccurate in their predictions of trade patterns between the United States and Mexico (Gallagher *et al.*, 2002). Such failures strongly suggest that the likelihood of being able to predict the kinds of macroeconomic shocks that largely determine actual levels of trade are quite small.

The USDA’s Economic Research Service (ERS) has used a computable general equilibrium (CGE) model to estimate changes in production of various agricultural commodities by country from different agricultural support scenarios in a multilateral trade agreement (Young *et al.*, 2001), but has not yet attempted to derive any estimates of changes in pesticide or fertiliser use from those data. However, the ERS has estimated changes in agricultural pollution in the United States that would result from the creation of a Free Trade Area of the Americas (FTAA) (FTAA Interagency Environment Program, 2000). The study uses a mathematical programming model of the US agricultural sector (the US Department of Agriculture's USMP model), combined with a geographic information system, to simulate changes in use of land and water resources from changes in trade flows. The model estimates changes in eight environmental indicators, including soil loss from water erosion, nitrogen and phosphorus losses to atmosphere and water, carbon fluxes and greenhouse gases.[3] Some analyses are also being conducted by members of the research advisory boards to the network of GTAP users. Fertiliser subsidies can induce a considerable increase in fertiliser demand by distorting the relative prices of agrochemicals and organic sources of nutrients, such as animal manure and sewage and thus discouraging the use of the latter (Runge-Metzger, 1996). Ending fertiliser subsidies has had an immediate and dramatic effect on fertiliser use. When New Zealand ended its subsidies to fertilisers in 1986 after two years' notice, fertiliser sales fell by nearly half. Fertiliser consumption in Czechoslovakia, Hungary and Poland fell by even greater percentages after fertiliser subsidies were ended there in 1990. Subsidy reform does not account for all of the reduction in any of these cases, but it is by far the most important reason (OECD, 1998b).

A demand curve for fertiliser can be constructed on the basis of historical data for both subsidised and unsubsidised prices, as has been done for Ghana for the 1980-89 period. On the basis of that demand curve, the decrease in fertiliser use from subsidy removal or reduction can be predicted at least in the short run. In most countries, the slope of the demand curve for chemical fertiliser depends on the fertiliser-product price ratios in the particular economy. An econometric study of 11 Asian countries shows that the demand for chemical fertiliser was relatively sensitive to the price ratio of rice to fertiliser — an elasticity between 0.4 and 0.7 in the short run, and even higher in the long run (Barker *et al.*, 1985). Thus, in countries in which new varieties and irrigation are sharply increasing, demand for fertiliser is relatively inelastic (Runge-Metzger, 1996; Hedley and Tabor, 1989). Subsidy elimination usually means that prices will increase by very large percentages, however, so even a very inelastic response translates into a very significant reduction in demand. In Indonesia, when fertiliser subsidies were removed in 1999, the price of urea increased by 150%, while paddy prices increased by only 50%. Although the elasticity of demand in response to this change was only -0.2, the result was a 30% decline in consumption compared with the previous year (Soedjais, 1999).

Longer-term responses to the change in relative prices, however, depend on the further evolution of the fertiliser-product price ratio. Increased world prices for crops can increase the use of fertiliser even after subsidy removal. After the New Zealand fertiliser sales fell sharply in response to subsidy removal, for example, they began to rise rapidly again in the early 1990s in response world market prices and by 1994 were back up to the 1983 level (OECD, 1998b). In most Sub-Saharan Africa the reduction or elimination of fertiliser subsidies in the 1980s, along with currency devaluations in connection with structural adjustment programmes, have made fertiliser much more expensive. Demand for fertiliser has varied from country to country, depending on how devaluations have affected prices for the major crops grown in the country. Export crops such as cotton and tobacco are heavily fertilised, whereas crops that are not traded internationally are fertilised much less. Thus Benin has increased fertiliser use ten-fold since the early 1980s, whereas Malawi has increased its fertiliser use by only 30% (IFPRI, 2001).

Sub-Saharan African countries present a special case in which the environmental impacts of agricultural subsidies may be positive rather than negative. In these countries, soils have suffered major losses in nitrogen and phosphorous nutrients in recent decades, and rates of fertiliser use are still only a small fraction of the global average. Fertiliser subsidies appear to be justifiable for these countries in order to reverse net nutrient depletion, reduce soil erosion and increase yields (Aune and Oygard, 1999; Warford, Munasinghe and Cruz, 1997).

Translating changes in fertilizer or pesticide use per hectare into changes in pollution levels is more complex. Cross-country and time series data for fertiliser use and agricultural land under cultivation are available from the FAO. The OECD has also compiled detailed estimates for pesticide use and nitrogen balances (the difference between input of application of nutrients entering the soil and the output or withdrawal of nutrients from the soil) for 22 OECD countries. Although total nitrogen balances at national levels are often used as a proxy indicator for nitrate pollution from chemical fertilisers and manure there are important differences between the two (Rørstad, 1999). The degree of nitrate pollution from a given nitrogen balance will depend on various characteristics of soil, climate and topography.

Aggregate national data on pesticide use is a less satisfactory environmental indicator than data on fertiliser use, because the thousands of pesticides in use have widely varying degrees of potency and toxicity. Nevertheless, Denmark’s Ministry of Environment and Energy (1998) has managed to construct an “Index of Load” taking into account the toxicity and relative importance of various pesticides in total pesticide consumption to show trends in the environmental load of agricultural pesticides from a reference period (1981-85) to 1996. However, this index would not be applicable to agricultural systems in which the profile of pesticide use is substantially different.

Data gaps and research needs

In addition to the continuing collection and reporting of data on agricultural support by OECD countries, the WTO Trade Policy Reviews are now the primary source of data on various types of agricultural support. A careful review, collation and analysis of support data in such reviews in recent years for countries not covered by the OECD system would appear to be a good basis for attempting to establish a broader disaggregated agricultural support database. Very little *ex post* analysis has been done to estimate the environmental impacts of a specific agricultural trade liberalisation agreement on specific crop or livestock sub-sectors. It would not require a major research effort to cover all the major sub-sectors for both NAFTA and the URAA. Such empirical research would provide a more solid basis for analyses of a possible future trade liberalisation agreement than now exists.

Systematic research and analysis on causal linkages between subsidies and environmental issues in agriculture has been focused almost entirely on OECD countries themselves. No studies have addressed the question of how subsidies in OECD countries affect the environment in specific agricultural sub-sectors of developing countries or how trade liberalization in such highly protected sectors as sugar or cotton would affect the environment in developing countries producing those crops. Such studies would need to consider the potential trade-offs between additional environmental stresses from increased production in developing countries and reduced production in highly subsidized OECD countries.

Irrigation water

Distinguishing characteristics of the sector

- Although irrigation subsidies could be considered as a subset of agricultural subsidies, its significance also transcends the agricultural sector. Irrigation accounts for 75% to 90% of total water use in developing countries (World Bank, 2001), and for more than one-third of water use in many OECD countries, so irrigation water subsidies can have major affects on water resource allocation.
- Infrastructure used in the provision of water for irrigation is often shared with other users of water, including hydro-electric works.

- Water is a resource that often cannot be used in agriculture without capital investments that are normally too large for individual communities to bear. By financing those investments, the government effectively confers a significant resource rent on a relatively limited group of beneficiaries.
- Incomplete property rights are pervasive, encouraging over-exploitation of ground-water resources on the part of irrigators who draw from the same aquifers (Tsur, 2000; Ringler et al., 2000).
- Water supplies have "lumpy" cost curves with discontinuities, and the supplier from a river-fed reservoir has high fixed costs and low variable costs as in a natural monopoly (Hall, 2000).
- With few exceptions, no internalisation of environmental externalities or of the resource depletion costs associated with irrigation is taking place through the pricing of irrigation water. The state's involvement in making irrigation possible, raises the issue of whether the absence of internalization of these costs confers additional benefits to irrigators and should thus be considered as a subsidy.

Defining and measuring irrigation water subsidies

Practitioners have defined a subsidy to the supply of irrigation water in two ways. The first, which is sometimes called the "cost-recovery" approach, defines it as public expenditures that benefit irrigators, net of the revenues from water charges paid by irrigators.[4] When small-scale irrigation works are involved, the measurement of these expenditures is generally straightforward (though data are often hard to obtain). Budgetary expenditures for each project are small, and therefore in the aggregate are less "lumpy" than for larger projects. Calculating gross costs attributable to irrigators for large (often publicly owned), surface-water-fed irrigation works is much more difficult, for several reasons. First, construction costs are often spread out over many years before the water is actually delivered to irrigators. Second, often the infrastructure and reservoirs are common to, for example, irrigation and other water uses, or irrigation and hydro-electric works. Third, governments usually levy some charge aimed at recovering some of these costs, but typically the pricing formulae employed attribute large portions of the project's costs to "public" uses (*e.g.* recreational boating) or to other private users (*e.g.* through cross-subsidies from electricity consumers), and use a rate of interest in the amortisation calculation that is below what a private entity would use. Calculating this degree of under-pricing on a project-by-project basis for a large country with multiple irrigation works of widely varying vintage would be an enormous task.

In addition to paying less than the full attributable costs of government-provided capital investments, irrigators are also supported through costs incurred by governments to cover the operation and maintenance of irrigation systems. These costs include the costs of personnel, materials and electricity for pumping, but also — in the view of some researchers — any foregone revenues that could have been earned by public utilities by running diverted water through hydro-electric turbines. Harden (1996) estimates, for example, that the opportunity cost to the federally owned Bonneville Power Administration (BPA) of diverting water from behind its Grand Coulee Dam hydro-electric plant to farms within the Columbia Basin Irrigation Project amounts to between USD 150 and USD 300 million a year. That is in addition to the opportunity cost to BPA of pumping the water up several hundred metres and charging irrigators only 1/28 of the retail price for the service. Special low electricity prices are also offered in some countries for private irrigators, mainly for pumping ground-water.

When it addresses cost recovery, the literature on irrigation water pricing generally does not include environmental and resources costs that are not otherwise internalized within the scope of costs that should be recovered. However, the European Commission has called, in a policy paper on water pricing policies, for the inclusion of marginal environmental and resources costs in the costs that should be recovered in the pricing of agricultural irrigation water, as well as water for household and industrial use (Commission of the European Communities, 2000). This implied that the Commission regards environmental and resource costs that are not recovered by the authority providing water as a subsidy to the user.

The cost-recovery definition of subsidies can be expressed either as an absolute value of net public spending or as the percentage of annual total expenditures not covered by revenues from water charges.

The second method of defining a subsidy to irrigators is based on the actual value of the water to the irrigator rather than the amount of public expenditure. According to this definition, an irrigation subsidy would be the difference between the water’s net economic benefit to the irrigator per unit and the charged price per unit. Economists consider this difference to be a “rent” conferred on irrigators by the water authority and have used this definition of irrigation water subsidy to understand the degree of distortion introduced into the agricultural economy by under-pricing the resource (Diao and Roe, 2000). The price that is based on the resource rent obtained from the water by the irrigator is also called the “shadow price” (Tiwari and Dinar, 2001; Lofgren, 1995). This value of the water to the irrigator is the marginal value product (MVP) of the water,[5] which is based on the incremental yield, or marginal physical product (MPP), of the water (Tiwari, 1998; Tiwari and Dinar, 2001). Ordinarily, the MPP is calculated based on a generic crop-water production function or on farm-level budget data. If both dry land and irrigated crops are produced within a homogeneous growing region, however, the MVP can be estimated by comparing the average commercial value of yield from irrigated lands in the district with the average yield on non-irrigated lands (Gardner, 1983). When water rights are traded, as in the western states of the US, the market value of the land incorporates the MVP of the water (Cummings and Nercissiantz, 1992). Land-value differentials thus provide yet another method of measuring the rent implicit in access to irrigation water and thus calculating the subsidy (Cummings and Nercissiantz, 1992).

Country-by-country irrigation water subsidy data available

No intergovernmental organisation or non-governmental organisation is now engaged in ongoing collection and reporting of data on irrigation water subsidies. However, some data have been collected using one or the other of the two definitions of irrigation water subsidy, by OECD, the World Bank and some independent researchers. The OECD has compiled rough estimates on the ratio of operations and maintenance costs as well as capital costs for 15 OECD countries. The percentage of operations and maintenance costs recovered by irrigation water prices (median or as ranges) have been published by the OECD Environment Directorate (1999) for various years from 1995 to 1998. These data are disaggregated by agricultural region for nine of the countries covered, and national averages are indicated for seven countries, with both regional rates and national averages given for two countries. These data suggest that most OECD countries were recovering operations and maintenance costs on irrigation systems, and that at least two (The Netherlands and New Zealand) were recovering some fraction of capital costs as well). They show that Greece, Italy, Mexico and Turkey were recovering only about 60 to 75% of operations and maintenance costs.

Eurostat, the statistical office of the European Communities, has adopted plans for documenting current cost and price levels for irrigation water in selected EU members, beginning with pilot studies in Luxembourg and Spain (Commission of the European Communities, 2000). A search of the Eurostat website did not turn up any additional documentation on these plans.

The World Bank has studied irrigation water pricing experiences in a number of countries, but has not done any systematic data collection on government spending on irrigation systems or on revenues from irrigation water charges. In a study of irrigation water pricing focused on five countries, Tsur *et al.* (2002) provided recent data on the proportion of costs recovered by irrigation water prices in South Africa, Mexico and Morocco, but provides no precise data on the ratio for Turkey and is unclear on whether or not "supply costs" include capital costs in China. The case studies do not provide data on actual budgetary expenditures on irrigation water systems in the three countries.

In its calculation of support to agricultural producers, OECD includes some estimates of government expenditure on on-farm irrigation works, operation and maintenance costs and, for a few countries, capital projects. However, the data are highly aggregate and not comprehensive.

Several studies have produced detailed data on estimated subsidies for several countries based on the resource rent definition. Saleth (1997) has compiled data on for 13 states of India for 1989-90 showing irrigation water prices as a percentage of net economic benefit, measured by the difference between incremental yield from irrigated lands and from non-irrigated lands. Diao and Roe (2000) report on data they have collected in Morocco comparing water's contribution to the gross value of irrigated agricultural outputs and the relationship between water charges and the value of those outputs in nine regional development authorities responsible for water resources management.

Bowen and Young (1985) have used a linear programming model to derive estimates of financial and economic net benefits to irrigation water supply for a case study area in the northern Nile delta of Egypt. Hussein and Young (1985) have estimated the net economic benefits of irrigation water for irrigators in Pakistan. Perry (2001) estimates the typical subsidy to irrigation water in Iran to be 90% of the full economic value of the water. In 1989, the national water authority in Mexico estimated the net economic value of water in northwest Mexico, and similar estimates of MVP were made in northern irrigation districts in the United States. In the early 1990s irrigation water subsidies in those districts were estimated at about 60% of the value of the water (Cummings and Nercissiantz, 1992).

Methodologies for estimating potential environmental impacts of subsidy removal

Under-pricing of water through irrigation subsidies has two kinds of impacts on the environment, both related to the excessive withdrawal of water:

the first is that, in regions in which irrigation would not have been profitable on some or all of the land without a subsidy, it artificially increases the area of land that is irrigated. The second is that it results in inefficient management of water on irrigated land (Rosegrant, 1997). When water charges are based on irrigated area, rather than on the amount of water used, they undervalue water resources. A study of various irrigation projects in Brazil reveals that the single most important cause of the water loss is excessive length of irrigation time (Ringler *et al.*, 2000), which is directly related to the lack of incentive to use the resource efficiently. Water tariffs based on area are by far the most prevalent worldwide. In a survey of farmers utilising 12.2 million hectares of irrigated land worldwide, Bos and Walters (1990) found that more than 60% paid charges on a per-unit-area basis, and that only about 25% paid water charges based on the volume of water used.

Prices based on volume used rather than on number of hectares irrigated can provide at least some incentive for irrigators to reduce water use by reducing the length of time that crops are irrigated, if the system can be enforced. According to a report by the US Embassy in Beijing, previous experiences in Northern China showed that simply shifting from pricing based on land area to pricing based on volume reduced total irrigation water withdrawals by 20% (Anon., 1997). But prices set at or near the MVP of the water should give irrigators the maximum incentive to reduce water consumption by adopting water-saving technologies, shifting to less water-intensive crops and reducing the amount of irrigated land (Lallana *et al.*, 2001; Varela-Ortega *et al.*, 1998). When the charges do approximate the MVP, the impact on water use is dramatic: Israel set the water price close to the MVP and achieved a 50% reduction in water use (Tiwari and Dinar, 2001).

All of these changes reduce the amount of water withdrawn, thus increasing instream flows of water and the amount of water remaining for other purposes. Using shadow prices to set water charges therefore reduces all the environmental impacts of the irrigation. However, it cannot actually reduce the salinization that has already occurred. Thus in places like the Aral Sea Region, where salinization has become acute, a tax on salt discharge is a much more effective means of reducing the further salinization of water resources than increasing the irrigation water tariff (Cai *et al.*, 2001a).

One study using a mathematical programming model (MPM) that specifies crop water requirements and water application costs, and calculating farm surpluses over a 20-year time horizon estimates the charges or combination of charges and bonuses that would be required to achieve 10%, 25% and 50% reductions in irrigation water consumption for six water districts in three Spanish regions (Varela-Ortega *et al.*, 1998). The study calculates the results of four pricing scenarios: a volumetric charge, a bloc-rate charge, a volumetric charge with a bonus for volume of water saved, and a block-rate charge with both penalty and bonus charges for quantities above and below 80% of the water allotment right. The simulation showed that the volumetric charge with bonus scheme cost the irrigators only half as much on average to achieve the 10% and 25% reduction goals and 34% less to achieve the 50% reduction. That suggests that combining optimum water charges with bonuses for water savings may be an effective way to influence irrigator practices. It also found that the technical endowments of the water districts (*i.e.* whether the district already has more efficient irrigation technologies) decisively influences both the degree of response to a given price increase as well as the impact of a given price increase on incomes.

The resource rent definition of subsidy provides an alternative approach to calculating the environmental consequences of subsidy removal. The price that approximates the MVP of the water represents the threshold price at which demand for irrigation water becomes elastic. Below that threshold, price increases will affect profitability but will not cause the irrigator to use any less water. However, prices at or near that threshold are expected to reduce or eliminate excessive water use (Tiwari and Dinar, 2001). The fact that prices for irrigation water have been far below the economic value of the water explains why researchers have consistently found irrigation water to be very price inelastic at lower prices but very price elastic above a certain threshold price (Lallana *et al.*, 2001; Varela-Ortega *et al.*, 1998). For example, OECD (1999c) reports data from institutional price simulations showing price elasticity of irrigation water demand in the Andalusian region of Spain at only -0.06 at the low end of the price ranges but -1.00 at "medium" prices ranges, and a simulation using a dynamic mathematical programming model showed an elasticity of -0.12 for the low ranges and -0.48 for "medium" price ranges for the same region.

Such calculations assume, of course, that volumetric pricing is both technically and economically feasible and politically possible. Many irrigation specialists argue that establishing volumetric pricing and raising water tariffs to the level necessary to bring about substantial savings in irrigation water would be too costly to make the "first-best" solution (even for full-cost recovery) a practical alternative, because of the large transaction costs necessary to establish volumetric pricing (Sampath, 1992; Tsur and Dinar, 1997; Spencer and Subramanian, 1997; Tsur, 2000; Perry, 2001; Tsur *et al.*, 2002). Its technical feasibility has also been questioned, given the existing infrastructure, management and regulatory frameworks (Perry, 2001).

Data gaps and additional research needed

Data on environmental indicators related to irrigation water remain very sparse. Data are needed on such indicators as rate of flow in watercourses, level of nitrates in water, soil toxicities and micro-nutrient deficiencies, level of groundwater table, and loss of productivity of land due to salinization at the water basin level both before and after the changes in subsidy levels. It is not clear from the available literature whether water authorities environmental agencies in the target countries or regions are already collecting any or all of these data.

Fisheries

Distinguishing characteristics of the sector

- In most fisheries, property rights are absent, and the fish remain a "common-pool" resource, even though a management regime may be imposing controls on both access and effort. Thus fishers generally have a stronger incentive to maximise production in the shortest possible time than would be the case with tradable quota rights. Systems of controls on catch and effort can mitigate but not eliminate this economic incentive problem.
- Fishing fleet capacity, defined as the maximum amount of fish that a fleet fishing in a particular fishery can catch in the absence of constraints on the availability of variable factors of production (Vestergaard, Squires and Kirkley, 1999; Lindebo, 1999), is an intermediate link between subsidies and environmental impacts. In theory, subsidies can affect fisheries resources by increasing fishing effort or fleet capacity, by reducing capacity or by slowing the reduction of that capacity.
- The non-malleability of fishing fleet capital is central to the over-capacity problem. Contrary to the assumption that capital in the fisheries sector can always move to a more profitable sector if overcapitalisation makes the fisheries sector unprofitable, fixed investment costs in the fisheries sector are so high relative to operating costs that vessel owners are very slow to respond to price signals (Munro, 1999; Munro and Sumaila, 2001). This characteristic of fisheries makes for a pronounced lack of symmetry in the environmental effects of introducing new subsidies to the sector, on the one hand, and removing subsidies to the sector, on the other. However, in all cases, the actual environmental effects of subsidies (including those intended to retire capacity) depend critically on the

effectiveness of the accompanying management system in limiting the catch.

- Uncollected resource rents are generally not a significant subsidy issue once a fishery is over-capitalized because resource rents tend to be dissipated (Clark and Munro, 1994). When a distant-water fleet gains privileged access to a coastal state's resources under a bilateral access agreement, while paying only a very small access fee, however, uncollected resource rents may represent a substantial subsidy to the producer (Porter, 1997).
- In most fisheries, the environmental costs of fishing are not internalized in the costs of fishing licenses (Milazzo, 1998).

Defining and measuring fisheries subsidies

Government financial transfers to the fisheries sector have a range of objectives and employ different methods to achieve them. The main method for estimating total subsidies to the fisheries sector is to aggregate all the financial transfers that alter the incentive structure of the fisheries sector by increasing revenues or reducing costs. Studies that have attempted to aggregate these data for OECD and APEC member states (OECD, 2000a; PricewaterhouseCoopers, 2000) have included the following types of transfers:

- direct payments to producer and processors from government budgets;
- transfers, including tax expenditures not specifically recorded in budget documents, that reduce the costs of fixed capital or of variable inputs;
- budgetary transfers for infrastructure or services that benefit the fishing industry or that are necessary to ensure that fishing is done sustainably;
- market price supports through trade measures;
- indirect financial transfers by distant water fishing countries to their own distant water fleets through payments to foreign governments for some part of the cost of access to their fishing grounds, which are usually treated as a subsidy to the cost of an input (WTO Committee on Trade and Environment, 2000; OECD, 2000a; Sharp, 2001); and
- general services.

Establishing an index of government support to the fishing industry for each country could be done by summing the annual values of all budgetary programs that benefit the fishing industry, adding an estimate of the annual value of price support to the industry and dividing by landed value of the fish catch for that year. Such a PSE-like index would not measure trade distortion but would be useful for maximising the transparency of support programs by allowing the relative support for the sector in each country to be compared with that of other countries. A complete accounting of subsidies in all OECD countries would require that price support be included. A few countries (Norway as well as Sweden and Finland before joining the European Community) have used price support measures to supplement other forms of support for fishing industries, and the EU has maintained a market price intervention program to protect fishers against low prices (OECD, 1993; OECD, 2000a). It appears, however, that price support has been sharply reduced in recent years in Norway (Milazzo, 1999; Flaaten and Wallis, 1999; OECD, 2000a). The inclusion of management services in calculation of the index for OECD countries as a whole in 1997 (without taking price support programs into account) increases the ratio of government financial transfers to the landed value of production from 4% to 17% (OECD, 2000a).[6]

Country-by-country fisheries subsidy data available

In 2000 the OECD published the most complete set of estimates of government financial transfers to marine capture fisheries. The study covers direct payments, cost-reducing transfers and general services, but not price-support, for all OECD countries for 1996 and 1997. It shows the total estimates as well as figures for seven types of support programs for each country. The data also include estimates of total support as a proportion of total landed value, but without taking into account price support. This allows comparison of countries in regard to the relative weight of subsidies. The omission of price support from the estimates means that they underestimate both total and relative support, at least for some countries. However, the OECD Committee on Fisheries is expected to undertake a separate study of price supports. The OECD study depended in part that governments supply some data that were not available as well as estimates for some types of support that are probably understated for some countries.

The OECD (2001c) updated the data on government support to the fisheries sector in member countries to include data on support programs in 1998 and 1999. The data for each member country were again broken down by types of programs. However, some countries (Belgium, Netherlands, Mexico and Poland) did not provide data on government transfers for either year, and Australia, Canada and Turkey did not provide date for 1999. Given the past levels of support by Canada and Mexico in particular, the lacunae in the 1998 and 1999 data could increase significantly the provisional totals for OECD countries for those years.

The second major international source of data on fisheries subsidies is the study commissioned by the APEC Fisheries Working Group (PricewaterhouseCoopers, 2000). It is accompanied by a detailed inventory of all identifiable programs, including infrastructure and management services, in 19 of the 21 APEC member states, including programmes of eight Chinese maritime provinces. Many programs reported, however, are not accompanied by any cost data, or the data are not specific as to the year being reported. A significant difference between the APEC and OECD data is that the former covers programs that support aquaculture as well as marine capture fisheries. Aquaculture subsidies account for 30% of the total of USD 12.6 billion in subsidies estimated by the authors, leaving a total of USD 8.9 billion for capture fisheries. In addition, the APEC study also includes subsidies to processing industries, which it puts at USD 0.7 billion annually. So the total of subsidies to the maritime harvest sector is estimated to be USD 8.2 billion. Despite the absence of tabulated national totals, and of any reported or estimated costs for many of the programs, reasonably credible national estimates for five non-OECD APEC countries (Chinese Taipei, Peru, Indonesia, Malaysia and Vietnam), ranging from USD 28 million to USD 279 million annually, can be extracted from the detailed inventory in the APEC study. The inventory also provides a more reliable estimate of Mexico's subsidies than what was reported to the OECD. On the other hand, China reported only USD 44 million in infrastructure-related programs at the national level, and Chile did not report any subsidy programs at all, both of which seem unlikely.

Milazzo (1998) constructed subsidy estimates that included off-budget programs — *i.e.* loans and tax breaks-for five major fishing countries (Japan, Norway, United States, Russia and China) as well as the European Community. Among the programs identified by Milazzo that were omitted from the other two studies is a Japanese loan-subsidy programme for which no figures are officially published. Milazzo found that the programme represented USD 3.7 billion annually, whereas the OECD figure for total Japanese assistance to investment and modernisation is only USD 26 million in 1996 and USD 21 million in 1997.

The USD 3.7 billion figure presumably represents the portfolio of subsidised loans being financed by Japan, and the figure of USD 26 million represents the annual total for actual interest subsidies. Technically, only the latter is the actual subsidy. In this case, the loan portfolio itself is probably a better guide to the impact of the subsidy than the interest subsidies themselves. Loans to fisheries investment can leverage a significant proportion of the new investment in a fishery, particularly when the industry is in financial straits, as it was in the 1990s. Much of the USD 3.7 billion portfolio of loans represents fishing investments that would not otherwise have been made. This is particularly true when the government has a history of forgiving fisheries loans, as Japan did in the 1990s (Porter, 1998c). It would also be important to know what loan guarantee programs Japan has operated alongside this portfolio of subsidized loans, because such programs also leverage much higher borrowing by the fishing industry (Milazzo 1998; Porter, 1998c).

Milazzo also noted that the United States pays for the cost of access so that its tuna purse-seine fleet may fish under a multilateral agreement with Pacific Island states, which are not financed by the Fisheries Service but by the Department of State. The United States does not acknowledge that these payments are a subsidy that is directly linked to its fishing rights under the agreement, so it has not been notified to the WTO or included in US reporting to OECD.

WTO notifications have so far provided data on only a small proportion of total fisheries subsidies programmes that should have been notified. Furthermore, many notifications do not even indicate their cost or value to the fishing industry (Schorr, 1998; Schorr, 2001; World Wildlife Fund, 2001). Even so, these WTO notifications have provided the only official documentation on certain off-budget programs, such as a Japanese program from 1991 through 1996 that granted an additional capital depreciation to fishing vessels beyond what was allowed in the tax code for other sectors. It was unclear from the way the programme was described in the notification, however, whether the amount (USD 4.2 billion in 1996) referred to the net value of the tax break to the fishing industry, to the capital investment qualifying for the measure, or to something else. Japan has stated in its comments on an earlier draft of this study that the USD 4.2 billion refers to the "total fishery production value of the fisheries that are covered by this tax scheme." This confusion surrounding this program underlines the need for greater clarity in the presentation of tax subsidies as well as lending subsidies in WTO notifications.

The World Wildlife Fund (WWF, 2001) has attempted to provide an overall picture of total subsidies worldwide by combining and correlating OECD, APEC and WTO data. Juxtaposing the two major studies and the limited WTO data and subtracting where needed to avoid double-counting is a useful exercise; but the adjusted totals from the three sources for 1996 and 1997 in the WWF study do not add up. WWF's reorganisation of the APEC inventory makes it easy to add up country-by-country totals for those programmes with quantitative values in the original APEC document. However, these totals are much smaller in most cases than the actual country-by-country estimates made by the APEC researchers, which were required in order to derive overall APEC totals by category of assistance.

The WWF study concludes that the one Japanese program of tax concessions to its fleets, which had never been reported to the OECD, and which does not appear in the APEC study, increases the amount of documented subsidies for global capture fisheries by more than 50%. WWF questions whether the combined figures now available come close to capturing the actual global total, suggesting that the total is at least USD 15 billion annually. As noted above, this total depends on including the USD 4.2 billion attributed to the Japanese tax subsidy program, which remains to be clarified.

Some additional data on fisheries subsidies are available from national, sub-national and EU websites. Steenblik and Wallis (2001) report on websites that provide detailed information on the support programs of Germany, the Netherlands, Portugal, Australia, Canada, Mexico, New Zealand and the United States. In some cases, these data may yield some details that are not covered in the OECD study; in others (*e.g.* Mexico and the US), they do not appear to cover all of the state's programs, some of which are administered by other agencies.

Methodologies for measuring potential environmental benefits of subsidy reform

Thus far, no methodology has been used to predict the impact of a change in the level or the distribution of different types of subsidies to a given national fisheries sector on the state of the fish stocks in the fishery. A recent FAO Expert Consultation (FAO, 2000) suggests two quantitative approaches aimed at estimating the impacts of subsidies on the sustainability of fish stocks: "dynamic mathematical modelling using real fishery data" and "econometric estimation of relationships based on time series, cross section or pooled data." The participants in the consultation noted the need to trace the effects of subsidies on costs and revenue and thus industry profits, and then to link changes in profits statistically to changes in fishing effort.

The FAO Committee on Fisheries agreed at its February 2001 meeting that the Fisheries Department should continue to investigate the nature and effect of subsidies on fisheries sector and called for a second Expert Consultation on subsidies. The current work program of the Fisheries Department calls for this consultation to focus on the impact of subsidies on the economic activities of recipients, based on empirical research using a common methodology. Meanwhile, the Fisheries Department itself is conducting surveys of the profitability of selected fisheries around the world, which are intended to help establish the role that subsidies play in profitability (FAO, 2001).

In the absence of experience with quantitative methods for linking the type and size of subsidy with changes in the level of fishing capacity in the fishing fleet, case studies offer a way of characterising the effects of certain types of subsidy under certain conditions. A number of case studies of fisheries subsidies provided by OECD countries and the European Community from the 1960's through 1980s illustrate the fact that, in fisheries that are still in the phase of rapidly growing capitalisation, the provision of subsidies, especially for capital costs, does have a pronounced impact on the rate of capacity growth (Porter 1998c; OECD, 2000a). In some cases, the relationship between subsidies and capacity expansion has been so close that the bulk of the capacity increase during a given period can be attributed to subsidies rather than to the effect of open access common pool character of the fishery. For example, Flaaten and Wallis (2000) found a strong positive statistical correlation between the level of interest transfers provided by the National Fishery Bank in Norway and the number of newly built vessels entering the fleet during the 1980s. By the second half of the 1990s, most OECD countries had redirected most of their financial transfers, apart from basic services, to the objective of capacity reduction.

It is not clear how many of the world's fisheries have reached the point at which total costs associated with fishing are greater than fishing industry revenues. Even after the equilibrium point has been reached, however, it appears that the perverse incentive inherent in the absence of property rights continues to push up or at least maintain the level of fishing effort. Standard economic models of the fishery (Gordon, 1954; Clark and Munro, 1975) were based on the implicit assumption that fleet capital is perfectly "malleable". In fact, however, fleet capital is relatively "non-malleable" — *i.e.* it cannot be easily adapted for use in another marine industry (*e.g.* freight transport) in response to price signals (Clark *et al.*, 1979; Munro, 1999). In addition, because of the high fixed costs of entry into the industry, vessel owners tend to remain in the industry as long as they can recover operating costs, even if they don't earn a satisfactory return on total investment (FAO, 1993).

As fisheries go from the stage of being under-exploited to the stages of being fully exploited and finally overexploited, the relationships among subsidy levels, the levels of fishing fleet capacity, and the state of fish stocks also change. Fleet over-capacity (defined as capacity above the level required for maximum sustainable yield) has existed in virtually every major fishing fleet for some years (Porter, 1998a). By the 1990s, the capacity of most major states' fishing fleets had begun to level off, and growth has continued to take place at a much slower rate compared with previous decades (Greboval, 1999).

In fisheries that already suffer from severe over-capacity, the primary issue in regard to the impact of fisheries subsidies is no longer whether they *increase* the level of over-capacity and overexploitation of resources, but whether they impede the process of adjustment to the economic conditions accompanying over-capacity. In general the removal of subsidies should increase the costs of fishing for the vessel owners in a given fishery, thus making unprofitable some vessels that were previously profitable. However, the very limited malleability of capital in the fisheries sector will limit the effect of subsidy removal on the level of capacity. In highly over-capitalised fisheries, even if subsidy removal does result in withdrawal of some vessels from the fishery, it is likely to remove only the least profitable vessels from its fisheries, and allow the remaining capacity to concentrate on the most profitable fishery. Thus subsidy removal will not necessarily alleviate the pressure on stocks (Vestergaard, Squires and Kirkley, 1999).

Case study evidence can also help assess whether and in what circumstances subsidies for the specific purpose of reducing capacity can bring about an improvement in the state of stocks or a lasting reduction in fleet capacity. Reviews of a number of case studies on subsidies for capacity reduction through vessel or license buy-outs (OECD, 1995; Gates *et al.*, 1997; Holland *et al.*, 1999; OECD, 2000a; Porter, 2002) indicate that they can reduce capacity in the short run, but that those remaining in the fishery tend to increase their capacity or effort, or both, in response, as long as the basic economic structure of the fishery remains distorted by the absence of property rights. The subsequent increases in capacity are often in the form of technological improvement rather than additional vessels.

The case study literature suggests that the impact of a given type of fishery subsidy on fish stocks through changes in the profitability of a given level of capacity and effort thus depend on the incentive structure and management characteristics of the specific fishery (*i.e.* whether the fishery is distorted by a “race to the fish” and how effectively the management system constraints catch and effort) and on the degree of over-capacity in that fishery. A matrix approach that takes into account all of the relevant characteristics of the fishery would facilitate the systematic assessment of the environmental consequences of subsidy introduction or removal (Porter, 2002).

Data gaps and additional research needed

Despite three overlapping major data sources (OECD, APEC and WTO), a few gaps in the data on financial transfers in significant fishing states remain to be filled. Tax subsidies and subsidised lending programmes could be better documented than they have been in OECD and WTO reporting. Major gaps also exist in the information reported for non-OECD countries. For example, data collected so far on Chile and for China appear not to be complete.

In order to use economic models or linear programming to establish the impacts of adding or withdrawing subsidies on economic decisions regarding fleet capacity and effort, researchers will need detailed data on fixed and variable costs to the fishing industry in various countries. Variable cost data need not be based on a survey of many fishing companies but can be compiled from random samples of vessels in the fishery, and vessel prices can be gleaned from public advertisements (Squires, Alauddin and Kirkley, 1994). A relatively small number of case studies in certain countries would make this research task more manageable.

In the search for evidence, either from empirical research or modelling exercises, that subsidy removal can bring about a reduction in the level of fishing effort, it would make sense to begin with cases that involve those combinations of subsidy types and fisheries most likely to demonstrate such a supply response. Thus the FAO Committee on Fisheries has decided to focus on cases such as subsidies to distant water fleets, the fisheries of third countries and under-exploited fisheries (FAO, 2001), where the sensitivity of vessel owners to a change in profitability is likely to be greatest.

Given the importance of subsidies for vessel buy-backs, more systematic work analyzing the record of past and present programs with a similar framework to assess the relationship between various conditions and results would make a valuable contribution to understanding the problem. Case studies that can be aggregated and compared could include updated information on programmes that have been previously studied. Particularly important is the adoption of a common methodology for gauging changes in fishing capacity and the health of fish stocks from the baseline to later years.

A major issue raised by fisheries economists regarding vessel buy-backs is the “moral hazard” problem. They argue that, if vessel owners have reason to believe that a first vessel buy-back program will be followed others in the future, they will adjust their behavior to take full advantage of the opportunity (Gates *et a*l, 1997a; Arnason, 1999; Munro, 1999, OECD, 2000a*;* Munro and Sumaila, 2001). This is an insight from economics which provides a strong hypothesis that should be verified in countries that have undertaken vessel buy-backs. No such empirical research appears to have been done.

Forests

General characteristics of the sector

- In much of the world outside the OECD, forests exploited by commercial logging companies are owned by states, which raises issues of resource under-pricing and resource rents.
- Roads and other infrastructure provided by states represent costs that logging firms would have to pay if the forests were on private land, either directly or through the fee paid to the owner for logging rights.
- Important linkages often exist between the processing sector and overexploitation by the production sector, through vertical integration and export restrictions on raw logs.

Defining and measuring forest subsidies

Three general types of subsidies have been widely recognised in the literature as having been provided to producers of forest products: budgetary subsidies for road-building or other services of value to the sector; resource rent subsidies inherent in provision of access to public forests at costs below the commercial value of the resource; and quantitative restrictions on timber exports or high log-export taxes, which benefit wood processing industries. Subsidies to the wood processing industry may be included in calculations of subsidy to the forest sector, because of the high degree of vertical integration between timber companies and sawmilling and plymilling industries in many countries and potential impacts on the state of the forest.

Net budgetary subsidies can be calculated by comparing budgetary outlays that benefit forest companies with revenues from those companies for government services. It is often complicated — though by no means impossible — to determine precisely what programs should be counted as benefiting the timber industry, as illustrated by the cases of the United States, Canada and Australia referred to below. Input price subsidies for the wood processing industry are estimated by calculating the difference between domestic log prices for wood processing industry and some reference price for the same logs. The subsidy from failure to capture full economic rents on timber is calculated as the amount of the "stumpage value" of the timber, or the value of the timber that is solely attributable to market demand for the good rather than to any cost of production, that is not captured by the state.

Resource-rent subsidies may be provided when states give concessions to logging firms to cut timber in state-owned forests and collect royalties that represent less than the commercial value of the timber in the concession. The resource-rent subsidy is calculated by subtracting the total cost of bringing the timber to market, including all forest charges and the cost of attracting the necessary investment, from the total stumpage value of the timber (Repetto, 1988; Day, 1998). In the case of Canadian softwood lumber, as many as four different methods have been used to calculate the stumpage value (Gale *et al.*, 1999). Some have suggested that one of the costs of production that should be included in calculations of stumpage value is the cost of forest regeneration, maintenance and protection (NIEIR, 1996; Ruzicka and Moura Costa, 1997). In many countries where regulation of forest concessions is weak, however, logging companies generally fail to carry out these basic services. Road-building is a cost of production, but would not be subtracted in calculating stumpage value if roads used by the logging firm are built and paid for by the state.

Country-by-country forest subsidy data available

No inter-governmental or non-governmental organisation has systematically collected data on government transfers to forest industries on a global or regional basis. However, some efforts have been made by government and non-government analysts to estimate these subsidies for certain countries. The World Resources Institute (WRI) has analyzed data on transfers to the forest sector in the United States. The WRI study asserts that the accounting methods used by the US Forest Service have systematically minimised its losses in selling timber from national forests to logging companies; accordingly, its analysis estimated that this program operated at an average annual net loss during the fiscal years 1993-1997 of USD 307.5 million (Sizer, 2000). Another independent study, using a cash flow analysis of Forest Service data (Oppenheimer, 2001), estimated the net loss in 1998 as being USD 407 million.

According to a critical study sponsored by the Sierra Club of Canada (Gale *et al.*, 1999), foregone budget and tax expenditures from the Canadian Government benefiting the logging and processing industries totalled approximately CAD 400 million in 1997. Transfers to the forest industry from the British Columbia Provincial Government for the same year, according to the same study, appear to have totalled about CAD 2.51 billion. The study may overestimate the level of subsidies by the Provincial government. Of this amount, CAD 1.73 billion was accounted for by the estimated resource rent subsidy, which is the median of an extrapolation from four quite disparate methodologies. The study lists ten separate British Columbia Provincial

Government programmes as subsidies, but only two of these appear to be actual transfers to the industry. The analysis includes all public administration costs of the Ministry of Forests and the Ministry of Environment, Lands and Parks, for example, as "forgone expenditures" (Gale *et al.*, 1999).

Australia's States have direct responsibility for forest management under the Australian Federal/State system. A 1996 study, sponsored by the Australian government (NIEIR, 1996) and aimed at estimating the total level of Australian forest subsidies, found that it was not possible to determine precisely how much of reported spending by State governments could be attributed directly to forestry operations, and how much had a public goods aspect. Based on an analysis of the State of Victoria alone, the study suggested that total Australian financial subsidies to forestry operations could have been in the neighbourhood of AUD 100 million, but might well be higher.[7]

Statistics on subsidies provided by the EU to the forest industries in its members states are incomplete, but a study of the 1994-2000 period estimated that the total of these subsidies was more than EUR 2.5 billion (*i.e.* EUR 416 million annually), and that half of that sum was spent on afforestation programmes (Toivonen *et al.*, 1999; Toivonen, 2001).

Finland has traditionally provided substantial budgetary support to its forest industry, and a recent study (Leppanen *et al.*, 2001) estimates that the level of financial support for that industry has been at roughly ECU 50 million annually, mainly in the form of grants for regeneration, since 1995. The same study estimates the "effective rate of assistance" (defined as the proportion of net assistance to the unassisted value added to the industry) at only about 1-2%. A compilation of studies on individual European countries (Ottitsch *et al.*, 2001) provides official data on government financial transfers to the forest sector in the Czech Republic, Poland, Slovenia and Estonia as of 1999-2000.

The only estimates for forest subsidies across a large number of countries are for resource rent subsidies. The most comprehensive study of the subject (Day, 1998) provides estimates of subsidies either in absolute values or percentages of total available resource rents - or both - for 17 tropical forest countries and two boreal forest countries (Canada-British Columbia and Russia). The data includes estimates for a date in the 1990s in nearly every case. However, the data is drawn from other published sources, and the author warns that the studies cited are not necessarily comparable in methodology and types of data collected. Some studies include fee evasion as part of the estimate, for example, while others do not.

Another study (Contreras-Hermosilla, 2000) reviews estimates for nine tropical forest countries, including three countries not covered in Day (1998), for years ranging from 1989 to 1997. In the country surveys, only Malaysia collected more than 30% of the potential rents. In addition to the 28 countries cited in these two studies, estimates of resource rent subsidies have been calculated for various years or periods from the 1970s to the early 1990s for Gabon (Repetto, 1988), the Ivory Coast and Guinea (Grut, Gray and Egli, 1991). Resource-rent subsidies in Peninsular Malaysia have been estimated for different periods by several authors (Vincent, 1990; Gillis, 1988b; Vincent and Hadi, 1993). Mohd Shawahid *et al.*, (1997) calculates that the State government has captured 20% of the total resource rent, but that it could capture 80% of that rent by using a tender system for allocating concessions. Estimates of resource-rent subsidies have also been developed for Canada for the early and late-1970s (Schwindt, 1987) and for the 1990s (Gale *et al.*, 1999). An Australian study estimated the under-pricing of hardwood and pulpwood logs through low royalty rates in two states (Marsden Jacobs, 2001). Thus estimates in regard to failure to collect all potential rents on the resource have been attempted for a total of 27 countries, although not, unfortunately, for the same years. Although it does not estimate resource-rent subsidies, a study by Vincent and Casteneda (1997) estimates resource rents for roundwood production in fourteen Asian countries, which could be compared with total timber royalties for the same countries to estimate resource rent subsidies.

Log export restrictions have been utilised as a means of supporting domestic wood processing industries in at least 13 countries since the late 1980s. An earlier report (LEEC, 1993) listed nine tropical timber countries that had imposed either bans or quantitative restrictions or high export taxes on log exports as of 1989. Scattered estimates of price-support subsidies to wood processing industries have come from case studies of log export restrictions or bans or high export taxes, or both, on raw logs in Canada, Indonesia, Malaysia, Ghana, Ecuador, Bolivia and Costa Rica. The U.S. Department of Commerce (1993) alleged that the log export ban instituted by British Columbia and some other timber-producing provinces was providing a subsidy to the Canadian softwood lumber industry estimated at roughly 8% of the value of Canada's softwood lumber exports. In the Indonesian case, the price of logs in the early 1990s, both under a complete log export ban and high export taxes that replaced it, was only about half the world price when sold to a processor independent of the logger but far less than that when the plywood operations were affiliated with the logging company (Varangis *et al.*, 1991; World Bank, 1993). Others have estimated that the Ecuadorian and Bolivian processing industries obtained logs at only 15% to 40% of what they would have paid in the absence of the log ban, and the Costa Rican processing industry could purchase logs at 18-52% of the world price (Kishor *et al.,* 2001; Simula, 1999).

Methodologies for estimating the potential environmental effects of subsidy reform

Researchers at the Finnish Forest Research Institute have published a study on the impact of public support for forestry on timber supply (Leppanen *et al*., 2001b), which may provide a research methodology for relating at least the rate of timber production to the level of support. Unfortunately, the paper could not be obtained during the time period of this study. No other methodology for measuring the impact of budgetary support to timber companies on forest health could be found in the literature.

As is the case with other sectors, subsidies are not the primary cause of unsustainable management of forest resources. The most important factor in the damage done to the forest by logging is the logging techniques used. And those techniques depend on other incentive measures, including greater stability and transferability of tenure and specific economic rewards for managing the resource for long-term sustainability, than on collection of adequate royalties (Paris and Ruzicka, 1991; Ruzicka and Moura Costa, 1997). Eliminating resource-rent subsidies, therefore, cannot by itself induce the concessionaire to exploit forests in a sustainable manner. Nevertheless, some resource economists have argued that the failure of governments to collect full economic rents on timber under-prices the resource to the logging firm and provides perverse incentives to log forests less efficiently than under adequate forest charges (Ruzicka, 1979; Repetto, 1988; Vincent and Binkley, 1992; Gray, 1996 and 1997).

Whether the collection of full resource rents can reduce the area harvested, or the intensity of the harvesting, however, has been the subject of intense debate. One view is that the extent of harvesting cannot be influenced by the level of resource rents collected, because cutting all the trees within the concession would still be profitable even without windfall profits (Day, 1998). However, other economists have argued that the imposition of adequate royalties on the logger increases the average and marginal cost of production, and that some trees with less favourable locations would become unprofitable to harvest (Ruzicka, 1979; Paris and Ruzicka, 1991; Ruzicka and Moura Costa, 1997).

The one empirical study of the relationship between royalty levels and cutting patterns (Amacher *et al.*, 2001) concluded on the basis of research in Peninsular Malaysia that harvesting rates on high-value species are more price and royalty elastic than are harvesting rates on low-value species. They estimate that the harvest on high-value species could increase as much as 5-10% for every USD 100 decrease in the royalty payment. Under-pricing of raw logs for domestic processing industries through log-export restrictions has the effect of reducing the efficiency with which the processing industry uses the logs as well as increasing its demand for the logs. In the Indonesian case, observers have estimated that the wood-processing industry was as much as 15-20% less efficient in turning raw logs into lumber and other wood products than the most efficient processors in Asia, meaning that 15-20% more trees had to be cut than would have been the case had the logs been processed elsewhere in Asia (Constantino, 1990; Gillis, 1988a). Similarly, the protected peninsular Malaysian processing industry has been assessed as consuming between 5% and 15% more trees per unit of sawn wood than unprotected competing processing industries (Vincent and Binkley, 1992). However, these studies have not made clear what empirical evidence supported the estimates.

Case studies have also shown that the under-pricing of raw logs results in a pronounced tendency toward over-capacity in the processing industry (Barbier *et al.*, 1995; Dean, 1995; Varangis *et al.*, 1993) because it transfers revenues from log producers to the wood-processing industry. Over-capacity in the processing industry by itself does not cause overexploitation of forests, but it is likely to increase the pressure on government to increase the total allowable cut. The artificial depression of prices of raw logs also depresses the supply of logs, but only if the harvesting and processing are not integrated. When the same companies control both harvesting and processing, as is often the case, the low prices for logs as domestic inputs also translates into greater supply for the processing industry. The higher the price-elasticity of demand for logs as domestic inputs, the greater will be the increase in the demand for logs (Dean, 1995).

The Indonesian case illustrates the effect on the rate of harvesting of under-pricing logs through a log-export ban. According to a study by an Indonesian NGO, the artificially low domestic prices of the logs and sawn wood had created significant overcapacity in wood processing industries, pushing processing capacity well beyond the maximum sustainable level of cut, which was then followed by unsustainable levels of logging (WALHI, 1991; World Bank, 1993). The upward pressure on harvesting levels was exerted not only by the price of logs but also by the absence of any effective control over concessionaires and the opportunity to capture export markets for plywood throughout Asia by underpricing competitors and then raising prices (Dauvergne, 1997). Ecuador's log ban has also been shown to have encouraged unsustainable rates of cutting by creating much greater demand for logs for the processing industry (Southgate and Whitaker, 1992). These qualitative analyses of the linkage between log export restrictions, over-capacity, and over-exploitation of forests cannot easily be translated, however, into quantitative research methodologies.

Data gaps and additional research needed

No comparable data on government transfers to the forest sector for similar years have been collected for OECD countries or for other groups of countries. The data that are available have not yet been consolidated in a single database, nor has any attempt been made to validate the data, nor to correlate it in terms of countries and periods covered. Nor has anyone compared the methodologies used to estimate forest subsidies based on program aggregation in the OECD countries. A synthesis and analysis of all the available data would be a useful exercise in the absence of a more systematic collection effort.

No quantitative methodology appears to have been used to estimate the environmental impacts of subsidy reform in the forest sector, except for the correlation analysis done by Amacher *et al.* (2001) to determine how rates of cutting of high-value and lower-value species change with royalty rates. That correlation is not quite the same as a correlation between resource rent-subsidy rates and cutting rates. It would be useful to explore whether the data exist to correlate different levels of resource-rent subsidy and cutting rates on high-value species or all species in the same forest areas. The inefficiency effect of export restrictions also remains to be studied systematically through collection and analysis of empirical evidence. This would require a researcher to obtain historical data from companies in countries with such export restrictions on the amount of timber consumed in ply-milling and saw-milling operations per common unit of output as well as similar data from companies in countries without export restrictions.

Energy

General characteristics of the sector

- Fossil fuels (coal, petroleum, natural gas) and electricity-generating technologies, are traded in regional or world markets, whereas most electricity is usually traded on domestic, but not international, markets.
- The competing energy sources have widely differing environmental implications, so the technology effect is one of the most important aspects of the environmental impacts of subsidies and their removal.

Defining and measuring energy subsidies

The International Energy Agency (1999) has defined an energy subsidy as any government action that:

- lowers the cost of energy production,
- raises the price received by energy producers, or
- lowers the price paid by energy consumers.

The IEA identified four major forms of energy subsidies: grants and credits (soft loans or interest-rate subsidies) to producers or consumers of energy; market price support (*e.g.* through regulatory requirements to purchase a given amount of fuel from a specific source at a regulated price or price controls to promote supply and consumption of particular energy sources); differential tax rates on different fuels; and publicly-funded research and development programs. Although these have very different impacts on energy markets, and on the environment, aggregating totals for these forms of subsidy provides a rough estimate of the magnitude of government intervention in the energy sector.

Several studies (*e.g.* OECD, 1997b) have focussed only on market price support to producers or, more commonly, on market transfers to consumers. In either case, they have measured the difference between actual prices and reference prices that would obtain in an undistorted market. This definition is not necessarily inconsistent with the first, but focuses only on the net effect of measures on the "price gap". Subsidies that allow producers to stay in business while selling their coal or some other product at a world price are not picked up. The reference price, which is a measure of the true market value of a unit of energy, is the opportunity cost of its consumption. It is represented by either the border price for internationally traded energy products or the cost of production for non-traded ones, adjusted for transport and distribution costs. The resulting estimate of the price gap is sensitive to the choice of exchange rate used to convert local currencies into a common currency. Both official exchange rates and exchange rates based on purchasing power parities have advantages and disadvantages, and both the OECD and the IEA have chosen to use official exchange rates in the past.

As originally conceived in the energy literature (Kosmo, 1987), the price gap method was applied only to measure subsidies that reduce the end-use price of energy, omitting from consideration those subsidies that actually raise the price of energy to the user. That was because the focus of much of the original work was on developing countries and countries in transition, where under-pricing of energy is rife. However, in developed countries certain high-cost energy industries have been protected from foreign competition. For example, in western Europe, coal producers who cannot compete against imported coal have been supported, primarily by requiring purchases at an official price that is significantly above the world reference price (and thus, in effect, taxing consumers of steam coal), supplemented by direct payments to producers (Steenblik and Coryonnakis, 1995). The OECD (2001d) calculated both subsidies that reduced end-use prices to below a world reference price and subsidies from consumers and taxpayers to producers that raised end-use prices above the reference price. The result was a country-by-country comparison in which “consumer price wedges” and “producer price wedges” were distinguished.

The calculation of price wedges is based on a reference price for fossil fuels that are traded internationally. For energy that is not traded internationally, such as most electricity and certain types of coal, a substitute reference price has to be constructed based on the cost of production. If data on the long-run marginal costs of production are not available, average production costs are estimated (World Bank, 1997).

Country-by-country energy subsidy data available

Estimates of support for coal are more systematic and complete than for other forms of energy. Indeed, the only regular, systematic reporting of energy subsidies carried out by an international body is the IEA’s annual estimation of PSEs for coal. These estimates were first produced in 1988, originally for five IEA member countries (IEA, 1988) — generally showing PSEs back to 1982 — and were later updated in the IEA’s annual review of *Energy Policies of IEA Countries* (1989-2001). More recently, the IEA’s annual statistical bulletin, *Coal Information* (IEA, 2002) has provided annual PSE estimates for coal production in France, Germany, Japan, Spain, Turkey and UK for 1991 though 2001, including calculations of aid per tonne of coal equivalent in local currency and in US dollars. It also distinguishes among specific subsidy programmes that benefit current production, programmes that do not benefit current production, and programmes to promote industry contraction for each of the six counties from 1990 through 2001.

The IEA also maintains a database on annual country-by-country expenditures for energy research and development for the years 1974-2000. Some partial information on subsidies is provided in its in-depth reviews of the energy policies of IEA member countries. Over the four-year period from 1998 through 2001, the IEA published detailed reviews of 25 IEA member countries. In 2002, the IEA also published a review of Russia’s energy policies. The reviews usually include data on some aspect of energy subsidies in the country. In some cases (Turkey and Russia, for example), the reviews provide data on support to the coal industry; in others (Spain, for example) they provide data on support for renewable energy technologies. None of these reviews appears to provide a complete analysis of energy subsidies provided by the country, however.

The European Commission's Directorate for Competition maintains a database on "state aid" (*i.e.* government transfers) to a limited number of sectors, including support provided by EU Member states for coal mining and for the general objective of "energy savings". Country-by-country data on these two categories of expenditure are provided for the years 1997 through 1999 in its second edition of *State Aid Scoreboard*.

Apart from these continuing exercises, the IEA, the OECD and the World Bank have all at different times tried to estimate market-price support to producers or market transfers to energy consumers in OECD or non-OECD countries. Based only on consumer price gaps, the World Bank (1997) estimated price-wedge subsidies for petroleum products, natural gas and coal for 17 non-OECD countries for 1990-91 and 1995-96. More-detailed data on which the calculations were based can be found in another study prepared for the World Bank (Rajkumar, 1996). These estimates of developing countries and former socialist countries in transition were presented only as "orders of magnitude", because the data on which they were based were of relatively poor quality.

Using the price-gap approach, the IEA (1999) developed estimates of energy subsidies (market transfers) in eight non-OECD countries (China, Russian Federation, India, Indonesia, Iran, South Africa, Venezuela, Khazakhstan) chosen because of their high levels of total energy consumption as of 1997-98. The studies found that energy prices in those countries were on average 20% below reference prices.

The OECD's Joint Working Party on Trade and Environment (OECD, 2001d) estimated both market-price support to producers and market transfers to consumers of fossil fuels as of 1996 for 26 OECD countries, as well as the average for all EEC member countries. These data represented weighted averages of price wedges for coal, natural gas, heavy fuel oil and light fuel oil. The same study also estimated fossil-fuel price gaps for Brazil, China, the former Soviet Union and India.

Ruijgrok and Oosterhuis (1997) estimated budgetary support for fossil fuels, nuclear energy, electricity, renewable energy and conservation for all

15 EU member states, Norway and Switzerland as of 1995. The authors quantify only direct payments and tax expenditures from state budgets that lower the cost of energy production, consumption or conservation, while acknowledging that indirect subsidies in the form of soft loans, provision of infrastructure and limiting liability for energy firms in the event of nuclear accident are also important. They find that 22% of these budgetary subsidies were going towards supporting renewable energy and conservation. The study does not attempt to relate the totals for direct subsidies to end-use prices or production costs, but it does compare the European countries according to the rate of subsidisation in dollars per TOE (tons of oil equivalent) of final energy demand.

Some governments provide data on budgetary expenditures for energy on websites related to their national budgets, which are useful to varying degrees in estimating total subsidies to the energy sector. Three such websites were found in the course of this study. Of these three, for Japan, provides one of the most convenient guides to its energy expenditures, showing the breakdown of its expenditures among measures to support domestic coal, oil development and stockpiling, new energy and conservation and nuclear power.

The Australian government's total expenditures on fuel and energy for fiscal year 2000-2001 and projections for such spending through fiscal year 2004-2005. New Zealand's expenditures for energy and for conservation and renewable energy resources are shown in the Detailed Statement of Appropriations of its 2000-2001 budget document on the website of its national treasury.

In two studies, the US Department of Energy's Energy Information Administration (EIA) calculated US subsidies to energy markets in the United Sates, based only on budgetary expenditures as of 1999. One study (EIA, 1999) analysed assistance to primary energy industries (including renewable energy and electricity); the other (EIA, 2000) analysed assistance to energy transformation and end use. The EIA identified nearly USD 4 billion in subsidies to primary energy in 1999, of which 60% were tax expenditures and virtually all of the remainder for research, development and demonstration. For energy transformation and end-use subsidies, it identified USD 2.2 billion, of which 63% was accounted for by direct expenditures.[8]

However, an independent study focusing on US subsidies to the oil industry alone (Koplow and Martin, 1998) calculated that tax expenditures benefiting the oil industry totalled USD 1.8 to USD 3.68 billion in 1995. Since tax expenditures for all forms of energy had fallen from USD 2.2 billion to USD 1.7 billion, according to EIA figures, the detailed comparison of the methodologies used in the two studies would be useful. Koplow and Martin estimated the total budgetary subsidy to oil alone, excluding defence expenditures that they relate to oil, but including the costs of the US strategic petroleum reserve, at between USD 4.5 billion and USD 10.9 billion.

A detailed study of Australia's direct budgetary and tax expenditures for fossil fuel production and consumption (Riedy, 2001) covers direct payments and tax expenditures and includes expenditures by state governments. The study includes some categories that would be more properly considered under transport as well as some that are arguably not transfers to industry or to consumers.

Methodologies for quantifying the environmental impacts of subsidy reform

One methodology for estimation of the impacts of energy subsidies would be to look at the impact of each type of subsidy on emissions separately, and then analyse any interactions among them. Steenblik and Coroyannakis (1995) and Newberry (1995) note that for coal, it is crucial to establish in the case of a particular subsidy whether the output is secured through purchase obligations, as ending those obligations could have the effect of reducing coal consumption through substitution effects, even if it contributes to overall higher energy consumption.

Several qualitative analyses of the effects of a reduction or removal of specific forms of subsidies to energy, summarised by Vollebergh (1999), support the generalisation that removing subsidies to the long-run marginal cost of a fuel technology are especially important to long-term emissions, because power-plant investment decisions are strongly affected by the relative prices of alternative technologies. They also provide evidence to show that withdrawing support to industrial consumers is more effective at reducing emissions than withdrawing the same amount of support to households.

Another method that has been used for economies in which the subsidy consists of a consumer price wedge, is to calculate the percentage change in prices that would occur with subsidy removal (derived by dividing the price gap by the reference price) and then using data on the elasticity of demand to estimate the change in consumption that would result from an elimination of the price gap (IEA, 1999). Price elasticities of energy demand vary, but a number of studies have found that long-run elasticities for energy demand tend to be around -0.5 (World Bank, 1997). The resulting estimate for reduced energy consumption can then be converted into estimates of reduced emissions by using relevant functions for carbon dioxide and other pollutants per unit of energy.

The environmental benefits of eliminating coal subsidies in Western Europe cannot be estimated on the basis of lowered coal prices but must include the impact of fuel substitution as well. In some of these countries (for example, the UK following energy-market liberalisation), the substitution effect can be strong enough to outweigh the effect of the reduced price of coal on pollutant emissions (Steenblik and Coroyannakis, 1995; Haugland, 1995). In Germany, however, coal consumers are already free to choose among different fuels, so the removal of support to the domestic coal industry would presumably lead them to switch to imported coal (IEA, 2000). It is the substitution effect from coal to less-polluting fuels that would represent the greatest gain in reduced local pollutants and greenhouse gas emissions. Thus cross-elasticities of coal and of competing fuels are crucial to the calculation of results.

The IEA (1999) estimated changes in carbon dioxide emissions for eight high-consuming non-OECD countries where domestic prices are on average 21% below reference prices. Comparing the baseline case and the case of an economy without energy subsidies in each country, the study estimates an average reduction in carbon emissions of 16% across the eight countries, with the reductions ranging from a high of 26% for Venezuela to a low of 8% for South Africa. However, the study was limited by the inability to estimate fuel substitution, based on cross-elasticities between the prices of different fuels, or the longer-run consumption savings from the more rapid development of energy-efficient technologies.

Most of the case studies in the environmental benefits of energy subsidy reduction have focused on greenhouse-gas emissions. However, several cases have estimated the reduction in acid emissions from elimination of subsidies to the electricity sector. These studies demonstrate that the benefits of subsidy removal for acid emissions can be proportionally greater than for carbon dioxide emissions in locations where acid emission are at levels that can cause environmental damage (OECD Annex I Working Group 1997).

Another model that has been used to estimate the impacts of a phase-out of coal subsidies is the C-Cubed model (Anderson and McKibbin, 1997). It utilizes a less complex model of world regions than the GREEN model, but has more economic sectors than the GREEN model and combines a dynamic macroeconomic modelling approach with a disaggregated, intertemporal general equilibrium model of the U.S. economy (McKibbin and Wilcoxen, 1996). In the study’s simulation, the phase-out of coal subsidies in Western Europe and Japan, plus imposition of a tax on the environmental damage from coalmining, would lower OECD carbon-dioxide emissions by 13% and global carbon emissions by 5%. If the major non-OECD countries were to remove subsidies to consumers by raising domestic prices of coal to world reference levels; moreover, it would reduce their carbon emissions by 4% and total world emissions by 8% below the baseline case.

The OECD (2001d) studied the environmental effects of a multilateral agreement on liberalisation of energy trade using the OECD's General Equilibrium Environmental (GREEN) multi-country, multi-sector model devised to quantify global costs of policies aimed at reducing carbon emissions. The model simulated three scenarios involving the elimination of all price wedges and consumer taxes (prices for consumers above the world reference price): one in which only OECD countries liberalise, one in which only non-OECD countries liberalise and one in which all countries liberalise. The simulation results showed that carbon emissions in the OECD countries would increase slightly by 2010 compared with the "business as usual" scenario if only OECD countries liberalise, but would be reduced by 6.2% if all countries liberalise and 6.3% if only non-OECD countries liberalise. The simulation had several weaknesses that probably caused it to underestimate the environmental benefits of energy subsidy removal: the model was based on incomplete data on producer price wedges, covered only subsidies to industry and power generation, excluded producer-price wedges for crude oil; and probably did not fully reflect the effect of technological improvements on energy efficiency, both through its assumptions and its 2010 cut-off date.

Regardless of the model used, in order to ensure that the environmental benefits of energy subsidy reduction are estimated as accurately as possible, the simulation must take into account the effect on the redistribution of production, the world price effect and the long-term effects of fuel substitution. Since subsidy elimination in higher-cost countries would redistribute coal production to lower-cost producing countries, and the environmental effects of that increased production must also be estimated. Little work has been done in this area. However, Steenblik and Coryannakis (1995) showed that, on a tonne-for-tonne base, a shift in production from deep underground mines in Europe to shallow surface mines elsewhere would considerably reduce emissions of methane from exposed coal seams. Coal subsidy removal would also increase the level of coal imports in response to more expensive domestic coal prices, raising world prices and lowering world-wide consumption through substitution and energy efficiency (Anderson and McKibbin 1997). That could be even more significant than the reductions in response to increased world prices in the countries covered in the study (OECD 1998*a*). Finally, the long-term effects of coal subsidy removal (*i.e.* more than 20 years) should be much larger than the shorter-term effects. Case studies that estimate results only to fifteen or twenty years in the future, such as Anderson and McKibbin (1997) and DRI/McGraw Hill (1997), are likely to underestimate total benefits, because the full effects would not be felt until existing plants are obsolescent and new investments are required (OECD 1998a; OECD Annex I Export Group 1997).

Data gaps and additional research needed

Energy sector price data are available for all OECD countries, but the data on producer price gaps even for major energy consuming non-OECD countries remains incomplete. Data on own-price and elasticity of demand and cross-elasticities are also still very quite inadequate for non-OECD countries. The IEA (1999), for example, had to rely on estimates for own-price elasticity that were often contradictory and included estimates that were outside plausible ranges.

Further simulations of multilateral agreements on phasing out all energy subsidies or coal subsidies alone are needed that incorporate more complete data on producer-price wedges, as well as the full long-term technological effects of fuel substitution in power sector investment.

Transport

Major characteristics of the sector

- The market for transport services is distorted by two basic structural characteristics: the significant elements of natural monopoly in the sector and its high ratios of fixed to marginal costs and high levels of sunk costs — costs that cannot be recovered by putting assets to alternative uses (ECMT, 2000; Roy, 1998).
- Competition between transport modes is a central feature of transport markets. Road transport dominates the transport market in advanced industrialised countries, accounting for 93% of all inland passenger-kilometres and 76% of all tons of freight kilometres in the ECMT countries in 1995 (ECMT 2000).
- Transport markets naturally fail in the direction of over-pricing and under-use of rail and under-pricing and over-use of roads (ECMT, 2000; Roy, 1998).

- Significant externalities in the road sector mean that marginal social costs can be far *above* average social cost. On the other hand, the very low level of externalities and increasing returns to scale in rail transportation means that marginal social costs for rail transport are far *below* average costs.

Defining and estimating transport subsidies

A transport subsidy could be defined either in terms of the gap between government expenditures to transport systems and the revenues collected from those systems (cost recovery) or by the failure to internalise external costs and other marginal social costs (congestion, scarcity, accidents, operating costs) in a specific mode of transport. Another way to characterize the differences between the two definitions is that one approach uses a "full allocated cost analysis", whereas the other uses a "short-run marginal cost analysis". A recent study (Sansom *et al.*, 2001) provides the clearest delineation of the differences in relevant cost and revenue categories used by each of the two approaches.

Which definition is relevant depends on the issue to be addressed. For efficient use of infrastructure, short run marginal social cost is the relevant basis for prices. Given the increasing returns to scale in railways such prices will not always cover total costs. On the other hand, in urban areas where the inherent economies of scale in road infrastructure are accompanied by the high external costs of road use, and especially congestion, such prices will much more than cover costs. This is because infrastructure expansion is constrained by competing uses for land in cities and a resource rent arises. Fully allocated cost analysis is of more relevance to optimisation of the supply of infrastructure outside urban areas, though cost recovery is not the only or necessarily most important, criterion in determining infrastructure supply. Social cost-benefit analysis is the relevant tool for guiding policy rather than a simple pricing rule.

The 41-member European Conference of Ministers of Transport (ECMT), as well as most transport sector specialists in Europe and North America, have adopted the principle that transport prices are distorted if they fail to internalise short run marginal social costs, including the marginal costs of maintenance, reconstruction and resurfacing in the case of roads, but not other producer costs or user costs that are internalised by transport users (ECMT, 1998; Litman, 1999; ECMT, 2000; Nash, 2000).

The implication of examining transport subsidies in relation to the optimisation of infrastructure use is that main instrument of policy for dealing with perverse subsidy in the transport sector is the application of Pigouvian taxes and subsidies to reduce externalities (Button, 1994) and maximise social welfare. Cost-recovery is not part of this framework.

A large number of external costs of transport could be considered in estimating the subsidy to a particular transportation mode. The ECMT has identified the costs of noise, vibration, air pollution, accidents, greenhouse gas emissions and congestion as most relevant to these calculations (ECMT, 2000). The external costs of infrastructure provision, such as land take, landscape impact and barrier effects on wildlife and human communities are often not considered because of data deficiencies. Differences remain over how to calculate costs related to greenhouse gases: some countries have adopted a "dose-response" approach to calculating those costs, whereas others have opted "shadow values" derived from the greenhouse-gas emissions reduction obligations adopted by governments or from the EU's carbon tax proposals (Ricci and Friedrich, 1999; Nelthorpe *et al.*, 1998).

Different valuation methods are used for different categories of external costs: for costs associated with accidents, revealed preference is the preferred method in Europe; for traffic noise, it is prevention costs. For air pollution, however, European researchers have developed a more detailed, bottom-up "impact pathway" approach that has been found to be more effective in estimating the external environment costs of a mode of transport. The "impact pathway" approach uses detailed site- and technology-specific data, pollution dispersion models, and detailed information on location of receptors and exposure-response functions and finally assigns monetary values to the identified impacts in order to estimate environmental impacts of energy-related activities. The ExternE model developed initially for air emissions from electricity production is the most notable example. It is has been used in a series of research projects focusing on calculating the external costs of transport in different countries or in particular international corridors in Europe (IER, 1997; Friedrich *et al.*, 1998; Vossiniotis and Assimacopoulous, 1999; TRIP, 2000).

In view of the amount of data and time required to implement the "impact pathway" approach to estimation, the research strategy in Europe has been to construct simplified functions describing the relationships between marginal external costs and certain parameters such as road type, vehicle technology and population density from a large number of case studies which can then be used, either by aggregation or averaging, for estimates at various levels (Ricci and Friedrich, 1999). The European Commission launched a new research project in 2000 aimed at upgrading emissions factors for all transport sources and providing consistent emissions estimates in all EU member states.

Given estimates of external costs, econometric models are available to compute optimum prices for each mode of transport in any particular city or country. An EU-funded research project, TRENEN II STRAN, developed econometric models that convert data on external costs, taxes and resource costs into optimum prices (Proost *et al.*, 1998). The difference between these prices and existing prices can then be viewed as the degree of perverse subsidy in regard to the city, transport route or country in question. These models suggest that perverse subsidies be focused primarily on the consumer price for car use in urban areas at peak periods, circumstances in which only one third to one half of the full marginal social costs are covered by the price. For off-peak travel and for buses, prices are much closer or even equal to social costs.

Defining and measuring the degree of subsidy by the difference between efficient prices and existing prices might be applicable at the national level for some small countries in which relatively few prices for individual areas or routes are required. These could be averaged based on the proportion of total VMT represented by each mode. But for countries with very large territories such as the United States, Canada or Australia, it appears to be impractical to attempt to estimate the subsidy based on efficient pricing criteria at the national level. Such estimates are only meaningful at the level of transportation routes or regions.

Country-by-country transport subsidy data available

Data on government spending on transport infrastructure costs and revenues from transport use across a number of countries were assembled in a 1994 study conducted for the International Union of Railway,s with the exercise repeated and published in March 2000. The data included in the study were estimated infrastructure costs for road (both passenger and freight) and for rail (both passenger and freight) for all European Union member states, Norway and Switzerland. Road network expenditure data for most of the countries in the study were lacking, however, and the estimates had to be based, therefore, on a total cost/total expenditure ratio of 1.3 from calculations for Germany and Switzerland and responses from other countries to questionnaires in the UIC study.

Based on various studies, the ECMT (2000) has estimated short- and long-run marginal costs of road and rail infrastructure use in Europe per 1 000 passenger kilometres or per tonne kilometre for several European countries.

The European Union is in the process of unifying its national transport accounts through a project called UNITE, which will provide national policymakers with the analytical tools and data for national transport accounts showing estimated external social costs for the country. The methodology for the national accounts and the marginal cost methodology for the project were established in November 2000. Pilot national transport accounts covering environmental costs and capital and maintenance costs for 1996 and 1998, are available for the UK (Sansom *et al.*, 2001), Germany and Switzerland (Linke, 2002) and should be available soon on the UNITE website for all fifteen members of the EU as well as Estonia, Hungary and Switzerland.

The European Environment Agency compiled and published figures on the external and infrastructure costs for rail and road transport for all 15 members of the European Community as of 1991 and their transport revenues for the same year, showing total social costs and total revenues in each case (EEA, 2000). The figures are presented misleadingly in terms of the "proportion of external and infrastructure costs covered by revenues". Though a more appropriate set of indicators has since been developed by the EEA they have yet to update the website.

The United States Department of Transportation's (US DOT's) Federal Highway Administration published a report in 1997 allocating highway-related costs, including all federal programs, to various types of vehicles. The report did not deal with air pollution, but an addendum published in 2000 estimated air pollution costs at USD 40 billion annually, which was about one-third lower than the EPA estimate for those same costs (US DOT, 1997; US DOT, 2000). The DOT report did not provide any estimate of global climate change-related costs from road transportation. It also considered that users internalise all congestion costs and two-thirds of auto accident costs.

The British government has published two relevant studies: Surface Transport Costs and Charges Great Britain 1998 (DETR 2001) which provides data on marginal social costs and revenues and also on fully allocated costs and revenues across the modes; Lorry Track and Environmental Costs (DETR 2000) which models road infrastructure and environmental costs for 16 categories of truck.

Methodologies for estimating environmental impacts of subsidy removal

Existing research methodologies referred to in the transport policy literature do not provide any means of measuring environmental impacts of subsidy reduction in the form of reducing the gaps in cost coverage at the national or sub-national level. Changes in the use of transport systems do not depend on aggregate changes in expenditures and revenues, but on the specific relationship between transport charges and particular transport routes.

In order to estimate the environmental impacts of perverse subsidies in the transport sector — or their removal — researchers must determine how much difference efficient prices for different modes of transport would make in total transportation demand and in modal shifts in relation to inefficient prices and then use emissions functions to translate the level of transport demand into pollutant emissions. Such estimates are based on gauging the price elasticity of transport demand under various circumstances. Demand for travel is almost always inelastic, but its price elasticity tends to increase if a high quality alternative mode of transport is available at lower cost (VTPI, 2001). In some regions, even inelastic demand responses to changes in the direction of efficient pricing could represent a significant reduction in environmental externalities.

In 1998-99, a consortium of European consultants and universities, with funding from the European Commission Directorate-General for Transport, produced the first comprehensive reviews in Europe of the empirical and modelling evidence of time and cost elasticities for car travel in the Netherlands, Italy and Belgium (de Gong *et al.*, 1999; de Gong and Gunn, 2001). The studies used three national transport forecasting systems (the Dutch National Model, the Italian National Model and a model developed for the Brussels region) to model the cost elasticities of number of car trips, number of kilometres travelled by car, number of passenger trips and car passenger kilometres for the three countries. It also modelled responses to prices of transport mode choice, including both short-term and long-term responses.

A modelling exercise focused on price elasticities of mode choice (Nash, 2000) carried out by an international partnership of European research institutions modelled the effects of different pricing scenarios in five transport routes based in 2010, including an efficient pricing scenario based on estimates of price-relevant social costs. The exercise compared the results of the efficient pricing scenario with the base case pricing scenario to identify the effect of efficient pricing on mode choice. A major limitation of the model used, however, was that it assumed a fixed total travel demand, so it could not estimate impacts of such pricing on overall demand.

The results of the study showed that the impact of the efficient marginal cost pricing scenario on modal splits depend very strongly on the characteristics of the transport route in question. In the study of Lisbon's traffic, the efficient pricing scenario (which involved price increases for cars of 38-80%) reduced private car demand by 20-40%, depending on time period and valuation of externalities, whereas demand for rail increased by 12-43%, depending on the same factors. Efficient pricing scenario, however, led to only a very modest increase in modal share for rail of 7-10% in the London-Paris and London-Brussels routes, and no modal shift in low-density and low-externality inter-urban passenger routes in Finland and from Oslo to Gothenberg. In the Oslo to Gothenberg route, efficient prices actually produced a shift away from less polluting modes of bus and train to the more polluting cars and air modes.

To assess the impacts of efficient pricing on mode choice and total passenger travel, USEPA can use transport price elasticity estimates and its MOBILE model (version 5a), or California's EMFAC model, which process data on vehicle trips, miles travelled, speeds and the fleet mix and emissions characteristics of the vehicle fleet to estimate emissions by type of pollutant (Deakin and Harvey, 1996). On the basis of modelling results for transport systems serving the San Francisco, Los Angeles, Sacramento and San Diego areas, Deakin and Harvey suggest that a VMT fee of USD 0.02 per mile could be expected to reduce VMT by 4.5-6.3%, carbon dioxide emissions by 4.8-5.7% and NOx by 4.3-5.4% from the 1991 baseline.

A Study by the University of California at Davis (Rodier, Abraham and Johnston, 2001) uses a regional transport and land-use model for Sacramento to evaluate a range of transport demand management policies. It simulated the imposition of a USD 0.05 per mile VMT fee, which was derived from the low-end estimate of external costs of automobile in Delucchi (1997), and average parking surcharges of USD 2 for work trips and USD 1 for other trips. The total charges represented a 30% increase in total driving costs above the base case and produced a 20% reduction in daily trips, a 21.8% reduction in VMT and reductions of 23%, 19.6% and 28.5% in carbon dioxide, NOx and particulate matter, respectively.

A number of empirical studies have investigated the sensitivity of vehicle travel to various types of road tolls. These studies indicate a price elasticity of –0.1 to –0.4 for urban highways, meaning that a 100% increase in toll rates reduces vehicle use by anywhere from 10% to 40% (VTPI, 2002). Those elasticities are nearly three times higher than the general estimate by Johansson and Schipper (1997) of long-run elasticity for annual mean driving distance per car in relation to taxation other than fuel (between -0.04 and -0.12).

The research results now available from modelling studies suggests that it is possible to go beyond rough orders of magnitude in calculating environmental impacts from a given change in prices for specific transport routes.

ENDNOTES

1. The transport sector is defined for the purpose of this study as ground transport, with the emphasis on passenger transport. It does not attempt to cover international air transportation systems or international shipping transport, which do not receive as much attention in the literature on subsidies in the transport sector.

2. Until recently, the OECD's Directorate for Agriculture, Food and Fisheries (OECD/AGR) referred to the PSE and CSE, respectively, as the producer and consumer *subsidy* equivalents. The term "subsidy" was changed to "support" mainly because it was felt not to be consistent with other economic terms that invoke an equivalency concept. However, some users of the concept outside OECD/AGR still use the original formulation.

3. This study appears as a chapter in *The Effects of the FTAA Agreement on US Agriculture*, published by the ERS in 2002.

4. One variant on the calculation of subsidy in terms of cost recovery for a set of irrigation systems covering a region or country includes only expenditures on the variable costs of the latest networks built (operations and maintenance as well as rehabilitation) but not the fixed costs (i.e. construction and depreciation costs).

5. It is, in effect, the price that a monopolist would charge if it could price-discriminate.

6. The possibility of constructing a PSE for the fisheries sector, as has been done for agricultural commodities and coal, was considered by the OECD Committee on Fisheries in the early 1990s but was rejected. It was argued that the technical obstacles to constructing PSEs for the fisheries sector would be too great and that the costs would outweigh the benefits because of the perishable and non-homogenous nature of fish products in the world market made it too difficult to establish reference prices by species for the purpose of calculating price supports (OECD, 1990; EC, 1990; OECD Committee on Fisheries, 1993a). Canada did not agree, however, that it was impractical to construct PSEs for fisheries subsidies, arguing that the value of price support through trade measures can be calculated for the entire fisheries sector of each country by dividing domestic sales by the sum of one plus the

tariff rate to provide an estimate of domestic sales that excludes the price increase provided by the tariff. The dollar value of the tariff measure can then be calculated by subtracting domestic sales excluding the tariff from actual domestic sales (Canada, 1990). By adding the total amounts of support from all three types of budgetary programs and valuing any loan guarantee and risk-reduction programs to the value of price support measures, it is possible to calculate a PSE for the entire sector.

7. Australia has indicated in comments on an earlier draft of this paper that it will provide updated information on financial transfers to the forest industry for the November 2002 OECD workshop on subsidies.

8. Several one-off studies of energy subsidies in the United States were also carried out in the early 1990s. The OECD (1997a), for example, published four studies that estimate subsidies broken down by programme; two of the studies had dramatically different high and low estimates.

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Australia
www.budget.gov.au/papers/bp1/htm/bs6-01.htm

European Environmental Agency
http://reports.eea.eu.int/ENVISSUENo12/en/page025.html

Food and Agriculture Organisation
http://www.fao.org/waicent/faostat/agricult/meansprod-e.htm#prices

India, Department of Fertilisers:
http://www.fert.nic.in/usfert/dfg9899.htm#2401

International Energy Agency
http://www.iea.org/statist/index.htm

Japan
www.mof.go.jp/english/budget/pamphlet/cjfc_q06.htm

New Zealand
http://www.treasury.gov.nz/budget 2000

Organisation for Economic Co-operation and Development
http://www.oecd.org; http://www.sourceoecd.org

World Trade Organisation
http://www.wto.org/english/tratop_e/tpr_e/tpr_e.htm

Unification of accounts and marginal costs for transport efficiency
http://www.its.leeds.ac.uk/projects/unite/index.html

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