

# Environmental Fiscal Reform

## PROGRESS, PROSPECTS AND PITFALLS

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# **Environmental Fiscal Reform**

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## ENVIRONMENTAL FISCAL REFORM – PROGRESS, PROSPECTS AND PITFALLS

This working document was prepared by Johanna Arlinghaus and Kurt Van Dender, with inputs from Luisa Dressler. All are at the Centre for Tax Policy and Administration, OECD. The work was overseen by Kurt Van Dender. The opinions expressed and the arguments employed herein do not necessarily reflect the official views of the OECD or its member countries.

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This working document is made available for discussion at the G7 meetings held in Bologna in June 2017. Comments can be sent to [Kurt.VanDender@oecd.org](mailto:Kurt.VanDender@oecd.org) and will be taken into account in the final version of the report.

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## 1. Introduction

Governments around the world face mounting environmental challenges, including the global problem of climate change and more local issues such as air and water pollution and waste management. Taxes are one instrument in the environment policy toolbox. They can be used to reflect the costs of pollution in prices, which encourages polluters to take account of these costs. This not only reduces pollution, but it also does it in effective and cost-effective ways. Taxes also raise government revenue, often a welcome property, in particular because taxing pollution (“taxing bads”) can often raise revenue at lower economic costs than taxing income or consumption in general (“taxing goods”).

Environmental fiscal reform, understood here as improved alignment of taxes and tax-like instruments with environmental damages coupled with socially productive ways of using revenues raised, has been introduced in several countries, to varying degrees. This report discusses the several aspects of, and experience with, environmental fiscal reforms. Section 2 introduces some terminology. Section 3 is an overview of the rationale for using environmental taxes, both from an environment and tax policy point of view. It also discusses two factors that have shaped environmentally related taxes to a considerable degree, namely concerns over loss of competitiveness of firms or industries when input prices rise as a consequence of higher taxes, and adverse distributional impacts or reduced affordability to households of some goods or services when taxes rise.

Section 4 of the paper is a “tour d’horizon” of the use of environmentally related taxes. It discusses how the amount of revenue raised through these taxes (Section 4.1), taxes on energy use (Section 4.2), taxes on road transport (Section 4.3), taxes and charges on waste (Section 4.4), taxes and charges for water abstraction and water pollution (Sections 4.5 and 4.6), and taxes on chemicals (Section 4.7).<sup>1</sup> For each type of tax, the weight of environment, revenue-raising and political economy factors on policy outcomes is briefly considered.

Taxes on energy use, including excise taxes and taxes on transport fuels, carbon taxes and taxes on electricity, raise the most government revenue among the set of instrument considered. Taxes and charges on road transport, including taxes on motor vehicles and congestion charges are the subject of policy debate but raise much less revenue. Waste and water charges contribute less to general government revenues, because levels are low but also because the revenues are often dedicated to specific purposes. They can and do play an important role in steering user and polluter behaviour. Taxes on chemicals (including on fertilisers, pesticides and other hazardous products) are potentially very effective but their revenue-raising capacity remains limited. Different rationales, underlie the introduction of these instruments, and shape their design.

The scope of the discussion is very broad. This report is a primer in environmental fiscal reform, meaning that the treatment of topics included is not very in-depth, and that several topics are not developed in any detail at all. The latter include, for example, the thorny issue of interactions between environmentally related taxes and other environmental policy instruments, detailed comparison of the upsides and downsides of the main market-based instruments (emissions trading systems and taxes, cf. Goulder and Schein, 2013, for a concise discussion), and the ‘twin topic’ of environmentally harmful subsidies. The latter is treated in a report prepared by the OECD as an input to Italy’s presidency of the G7, similar to this report. Detailed country analysis is absent from the text too. For a very recent example, see the report on France by Pourquier and Vicard (2017).

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<sup>1</sup> Due to their limited practical relevance and revenue-raising capacity, taxes on other hazardous chemicals, taxes on single-use bags and packaging are not discussed in this paper.

## 2. Some notes on terminology

This section provides definitions of some terms that are frequently used in discussion of environmental fiscal reform

**Market-based instruments** are one of several categories of environment policy instruments. They “[...] seek to address the market failure of 'environmental externalities' either by incorporating the external cost of production or consumption activities through taxes or charges on processes or products, or by creating property rights and facilitating the establishment of a proxy market for the use of environmental services.” (OECD, 2007). This definition refers to “internalisation of external costs”, which makes sure that the damage caused by pollution is reflected, or at least better reflected, in market prices. Market-based instruments are different from regulation-based approaches to reducing environmental damage, as the latter do not directly modify prices, even if compliance of course is costly in general. Section 3 explains why price-based instruments tend to reduce pollution at lower costs than regulations.

**Environmental taxes**, a subset of market-based instruments, are taxes “whose tax base is a physical unit (or a proxy of it) that has a proven specific negative impact on the environment. Four subsets of environmentally related taxes are distinguished: energy taxes, transport taxes, pollution taxes and resources taxes” (OECD, 2005). Tradable pollution permit systems, henceforth emissions trading systems, similarly put a price on processes or products with a proven negative environmental impact. It may be worth noting that the definition of environmental taxes does not explicitly link the tax to the size of the environmental damage, or the external cost, but instead only refers to the tax base. Nevertheless, environmental taxes are often implicitly understood to be taxes that aim to improve alignment of tax rates with (marginal) external costs.

**Environmentally related taxes** are defined as “any compulsory, unrequited payment to general government levied on tax-bases deemed to be of particular environmental relevance” (OECD, 2004). Here too, there is no explicit connection to external costs. That is to say, environmentally related taxes “particularly affect the environment” whether this is the policy intention or not, and whether or not tax rates align with external costs or not. In the remainder of this report, the term ‘environmentally related taxes’ will be used except when there is reference to marginal external cost pricing alone. This term is preferred because alignment with marginal external costs is one of several potential policy objectives that influence tax rates, meaning that referring to environmental taxes is potentially too narrow to describe policy practice now and going forward.

**Environmental fiscal reform (EFR)** is defined in several ways: In one definition, EFR refers to “a range of taxation or pricing instruments that can raise revenue, while simultaneously furthering environmental goals. This is achieved by providing economic incentives to correct market failure in the management of natural resources and the control of pollution.” (IBRD – World Bank, 2005). In this view, EFR is little else than the application of environmentally related taxes. A second definition sees EFR as “a tax shift from labour towards environmental use, supplemented by the reform or removal of environmentally adverse subsidies” (EEB, 2017). This definition is more specific on revenue use, including a requirement that revenue from environmentally related taxes be used to reduce labour taxes, and that environmentally harmful subsidies should be reduced or eliminated. An intermediate view is as follows: “EFR is frequently discussed as a means of bringing about a so called ‘tax shift’ in which a progressive increase in the revenues generated through environmentally related taxes provides a rationale for reducing taxes derived from other sources, such as income, profits and employment, the taxation of which is less desirable. [However], even where there are no explicit offsetting reductions in other forms of taxation, fiscal consolidation through increasing environmental tax revenue might implicitly keep the level of other taxes below that which might otherwise have prevailed”.

In the present paper, EFR is understood much in the same way as in the third definition. EFR involves: (a) environmental policy using market-based instruments to reflect the cost of environmental damage in prices faced by polluters and (b) raising public revenue and deploying it in a socially useful way. As is argued below, productive revenue use can mean different things in different circumstances. Requiring productive revenue use is not an empty statement, however, as it highlights one of largest potential drawbacks of the use of revenue-generating instruments, namely that the revenue would be squandered.

### **3. Rationales for using environmentally related taxes and factors shaping their design<sup>2</sup>**

Improving environmental outcomes and raising revenues are usually the prime motivations for introducing environmentally related taxes. Compared to other environmental policy instruments, environmentally related taxes and market-based instruments more broadly have the key advantage that they can improve environmental outcomes in a cost effective way, while also raising government revenues (Section 3.1). Section 3.2 discusses how environmentally related taxes can contribute to broader tax policy, which has the objective of raising revenue in a way that maximally contributes to economic efficiency and to equity. Environmentally related taxes in practice are often shaped by what are often called ‘political economy considerations’, namely competitiveness of firms and sectors, and direct equity impacts. These concerns are discussed in sections 3.3 and 3.4 respectively, with the main insight being that competitiveness and equity objectives, where relevant, are better addressed by flanking measures than by tweaking the rates of environmentally related taxes.

#### **3.1 *Improve environmental outcomes***

Without government intervention, there is no market incentive for firms and households to take environmental damage into account in their production or consumption decisions. Firms and households are not necessarily indifferent to pollution, but in most cases the cost of environmental damage they cause themselves is spread across many people and has little or no direct effect on them, so their incentive to cut pollution is very small or absent. The protection of the environment therefore generally requires collective action, usually led by government, through taxes or regulation. This section motivates why market-based instruments are superior to regulation in many instances, and compares taxes to emissions trading and tax incentives.

Improving environmental outcomes is often a prime motivation for introducing environmentally related taxes or implementing environmental tax reforms. Environmentally related taxes can achieve the improvement of environmental outcomes in a relatively cost-effective way, providing economic actors with the flexibility to adapt to the instrument in their preferred way and providing dynamic incentives to innovate. They also avoid having to design complex markets (in contrast to trading systems), and do not risk having to pick winners and subsidising existing behaviour (in contrast tax incentives).

As a rule of thumb, the objective is to align taxes more closely with marginal external costs. This is a pragmatic way forward even when these external costs are not precisely known or when other policy objectives would in principle call for some degree of deviation from marginal external costs. It is assumed that interactions with other taxes or prices are not sufficiently strong that they reduce the case for marginal

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<sup>2</sup> The discussion in this section builds on existing OECD and related work, updated for new information and insights. OECD (2011) is an important source, as is more recent OECD work on the effects of carbon prices on competitiveness, the effects of energy taxes on the distribution of income, and their impacts on the affordability of energy. Some reference is made to forthcoming OECD taxation working papers, on revenue use and the impacts of permit allocation rules on investment incentives.



external cost pricing.<sup>3</sup> Alternatively, taxes could be chosen to attain policy-defined levels of abatement (Baumol and Oates, 1971).<sup>4</sup> In practice, most environmentally related tax rates are far below marginal external costs, and also below levels to achieve significant behavioural changes, so that moving in the direction of alignment through tax increases most often is a sensible approach.

### *Choosing between market-based instruments and regulation*

Environmental policy typically has been, and to a high degree still is, dominated by regulations. These approaches can be prescriptive and highly targeted – *e.g.*, banning or limiting particular substances or requiring certain industries to use specific technologies – or they can be more flexible – *e.g.*, setting sales-weighted fuel economy standards for passenger cars. Over recent decades, interest has grown in using market-based instruments such as taxes and tradable emission permits, in addition to or instead of regulations. There are a number of reasons for the increasing use of environmentally related taxes.

Taxes can directly address the market failure that causes markets to ignore environmental costs. A well-designed environmental tax increases the price of a good or activity to reflect the cost of the environmental harm that it imposes on others, or at least moves the price in that direction. The cost of the harm to others, which was external to markets, is then internalised into market prices. This ensures that consumers and firms take these costs into account in their decisions. In essence, environmentally related taxes modify relative prices so that consumers include environmental costs in their spending decisions more accurately.

It is useful to note that it is assumed that moving in the direction of external cost pricing is a sensible policy strategy given the broader regulatory and fiscal context in which an environmental tax is introduced. This means that it is assumed that the broader context is sufficiently conducive to taxes being able to trigger the intended behavioural reactions. Such policy alignment, however, is not to be taken for granted entirely, and it is worthwhile to investigate the broader tax and regulation system to ensure that the power of environmental taxation can fully be harnessed (see OECD, 2015d, for a discussion of policy alignment in the context of climate change).

Regulatory approaches often include specific instructions on how to reduce emissions or who should do the reduction. Similarly, subsidies and incentives for environmentally preferable goods or practices involve the government steering the economy in favour of certain environmental solutions over others. In both approaches, the government is trying to “pick winners” – directing the market in a prescriptive way. This requires significant information about ever-changing conditions and technologies (which can be expensive to obtain or may simply not be available), and carries significant risk of making suboptimal choices.

Regulations generally result in abatement costs at least as high as those that would result under taxes, since they force particular types of abatement, even if cheaper alternatives are available. As long as environmental problems and the solutions for them are broadly and well understood, this information asymmetry may not be too much of a problem, and regulation can work well. This may have been the

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<sup>3</sup> For example, if a commuter can only get to work by car, and if their labour income is already heavily taxed, it may not be efficient to introduce a toll even when congestion is strong. If the toll revenue were used to cut labour taxes, the net tax burden would not change. If it were used for something else, the labour market inefficiency would increase. However, in practice congestion tolls can address many other behavioural margins than the stylised example allows for, *e.g.* trip timing, mode choice, which renders the toll useful if revenue recycling avoids a higher labour tax burden.

<sup>4</sup> Among the policy areas surveyed in this report, it appears that the Pigouvian approach to determine tax rates or price levels receives most attention in the discussions on climate change, and debate on the taxation of waste or chemicals is more strongly shaped by the tax rates necessary to achieve desired outcomes.

situation in early phases of environment policy, but as the easy and cheap abatement options have been exhausted, it becomes more important to let the decision on how to abate to those best informed about the available options and their costs, namely the polluters.

The higher cost of the polluting activity that results from the environmental tax makes the activity less attractive to consumers and businesses. In contrast to regulations or subsidies, however, a tax leaves consumers and businesses full flexibility to decide how to change their behaviour and reduce the harmful activity. This allows market forces to determine the least-cost way to reduce environmental damage.

For example, many countries impose significant taxes on motor fuels like petrol and diesel to raise revenue but also because the use of these fuels contributes to global warming and local air pollution. The resulting increase in the cost of driving a vehicle is an incentive to reduce emissions that could be achieved in a number of ways, in both the short-term and the long-term:

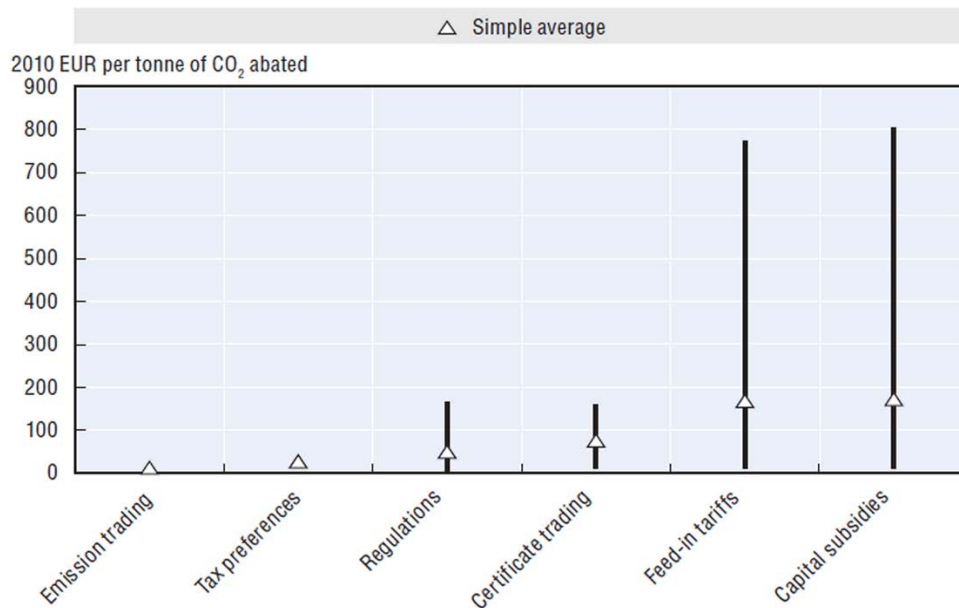
- Drive a smaller or otherwise more fuel-efficient vehicle.
- Drive a vehicle that uses a lower-emission power source, such as a hybrid-electric vehicle.
- Drive less, perhaps by greater use of low- or no-emission alternatives like public transit, cycling, walking, living closer to the place of work, or otherwise changing habits to reduce the need to travel.

The environmental tax provides a greater range of abatement options than instruments such as a regulation requiring a minimum fuel efficiency level for vehicles or a subsidy that privileges electric vehicles, which target only some solutions. Of course, if regulations are tough enough and strictly enforced, they can have significant effects. However, this achievement may be bought at the expense of unnecessarily high costs. For example, for households that do not drive very much but do not have the option of switching to different transport modes, investing in higher fuel efficiency is very costly per unit of service they will get from the vehicle. Such households may be better off by paying the fuel tax for the trips they cannot cut. Fuel taxes allow these diverse responses that reflect the specific profile of each polluter. This feature becomes increasingly appealing as the profiles of polluters differ strongly and the abatement costs become high.

The high cost-effectiveness of market based instruments compared with other environmental policy instruments is not just a theoretical idea. There is strong evidence to support it. To give just one example, the 2013 OECD report on *Effective Carbon Prices* calculated the abatement cost per tonne of CO<sub>2</sub> that is associated with a range of instruments in different sectors. The following two figures, taken from the *Effective Carbon Prices* report show the abatement costs in two sectors averaged across the range of countries studied. In the electricity sector, emissions trading and tax preferences cut emissions at very low cost. Feed-in tariffs and capital subsidies cut emissions too, but at a much higher cost per tonne, approaching EUR 100 average in the case of capital subsidies. In the transport sector, motor fuel taxes abate carbon at EUR 55 per tonne. Fuel mandates and capital subsidies result in much higher abatement costs, more than EUR 400 per tonne on average. In extreme cases, the abatement costs reach EUR 1000 per tonne.

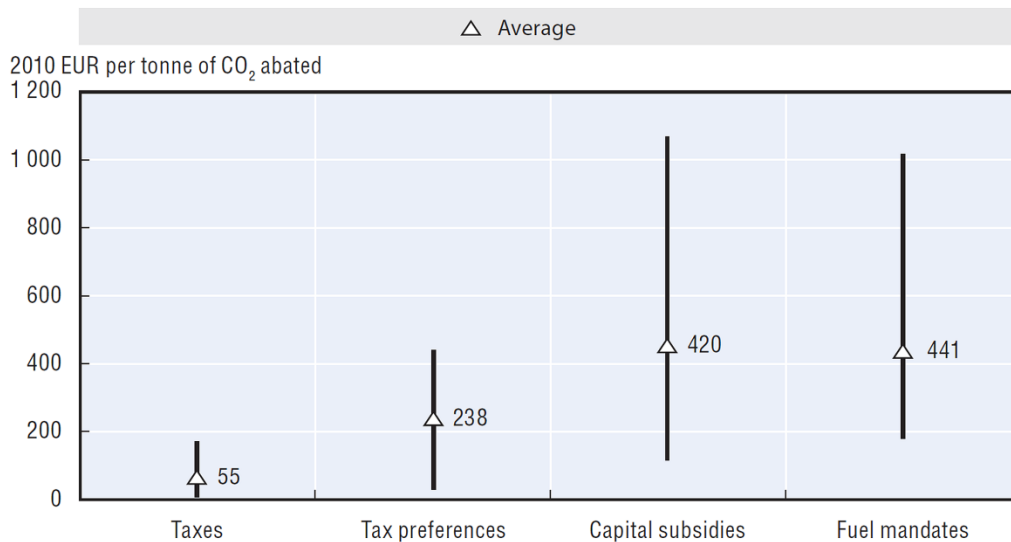
It can of course be argued that the instruments that are being compared do not exist exclusively to cut carbon and should not only be considered as promoting static efficiency but also dynamic efficiency. A broader assessment could arguably be more favourable towards some of the non-tax or trading instruments considered, e.g. where learning effects and market development concerns come into play. Nevertheless, it is clear that taxes manage to promote cheap abatement in a static context, and that they are part of the efficient set of dynamic policy instruments.

**Figure 1: Estimated effective carbon prices in the electricity sector, by instrument category**



Source: OECD (2013), *Effective Carbon Prices*, OECD Publishing, Paris. DOI: <http://dx.doi.org/10.1787/9789264196964-en>  
 Note: the height of the bars represents the range of effective carbon prices estimates found for the different instrument categories; the triangles represent a simple average of these estimates. "Regulation" refers to renewable portfolio standards.

**Figure 2: Estimated effective carbon prices in the road transport sector, by instrument category**



Source: OECD (2013), *Effective Carbon Prices*, OECD Publishing, Paris. DOI: <http://dx.doi.org/10.1787/9789264196964-en>

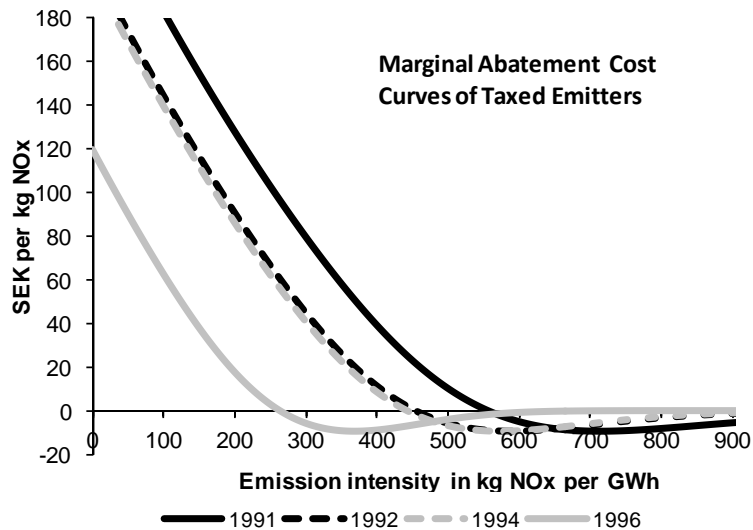
The flexibility of response associated with environmentally related taxes also provides other benefits:

- Ongoing incentive to abate.* A target-based or technology-based regulation provides no incentive to abate once the target or technology standard is met. By contrast, environmentally related taxes provide a continuous incentive to abate at all levels of emissions, even after significant abatement has already occurred. Figure 3 shows how the NO<sub>x</sub> tax in Sweden has (arguably) contributed to the reduction of abatement costs per unit

of emissions, over time, and OECD (2013b) shows that regulated plants have further reduced NOx emissions intensity per unit much further in subsequent years.

- *Reduced need for support of low-emission alternatives.* Environmentally related taxes increase demand for low-emission alternatives, like public transit and cycling in the case of taxes on automotive fuel. This can result in economies-of-scale that help to make such alternatives more viable, without a need for direct subsidies.
- *Strong incentive to innovate.* Taxes increase the cost to a polluter of generating pollution, providing incentives for firms to develop new innovations and to adopt existing ones. In the example above, the increased demand for more fuel-efficient and alternatively powered vehicles induced by fossil fuel taxes provides an important incentive for automakers to develop such vehicles and for consumers to buy them. Under regulation-based approaches these incentives disappear once firms have complied with the regulated standard. Enhanced innovation lowers the cost to society of addressing environmental challenges in the long run.

**Figure 3: Marginal abatement cost curves for NOx emissions Sweden, 1991, 1992, 1994, and 1996**



Source: Höglund-Isaksson (2005)

*Choosing among market-based instruments: taxes, trading or tax incentives*

Environmentally related taxes modify relative prices to reflect environmental costs. This can in principle also be achieved by making less damaging choices cheaper, so an alternative to taxing environmental “bads” is to provide tax relief or other subsidies for environmental “goods”. The tax system can be used to subsidise environmentally beneficial goods or actions by, for example, VAT exemptions for energy-efficient appliances or favourable depreciation rates for capital investments in renewable energy or pollution abatement (see Greene and Braathen, 2014, for an in-depth treatment). Like other subsidies, however, tax incentives have a number of important limitations:

- Since it is difficult to subsidise all potential environmentally beneficial alternatives to the harmful activity (some of which may not exist yet), tax subsidies inevitably involve “picking winners”, which may disadvantage other good alternatives. For example, unlike a tax on road fuel, a subsidy for low-emission vehicles does not provide any incentive for commuters to consider alternative forms of transportation, such as public transit or cycling. In general,

compiling the list of items or of characteristics that define eligibility for tax incentives is difficult, and administering the tax incentive becomes more challenging as more effort is made to target the tax incentive, implying a trade-off between effectiveness and administration costs.

- By reducing costs, tax subsidies may indirectly increase pollution. For instance, unlike a tax on vehicle emissions or road fuel, a subsidy for hybrid electric vehicles may encourage people to drive more.
- Tax incentives are frequently found to provide subsidies to actions that would have been taken in their absence while resulting in limited additional investment. They often also accrue disproportionately to more affluent households. Also, it has been observed that tax incentives result in higher supply prices (e.g. with inelastic labour supply, tax incentives for R&D can result in higher wages for R&D workers) instead of, or in addition to, increased adoption of the intended behaviour.
- Tax incentives are a form of spending public revenue, in contrast with taxes, which add to public revenue. This tends to favour the use of taxes, as government revenue usually is scarce.

These limitations of tax incentives do not imply that they should not ever be used, but do suggest caution, and that they likely should be used less frequently or designed more strictly than currently is the case. Where there are positive externalities — i.e., where markets provide too little of an activity compared with what is socially useful — tax incentives can be the policy instrument of choice. Support for research and development (R&D) is an example of where positive externalities, or spill-overs, exist. The case for support for green R&D is even stronger than for R&D support in general, on the grounds that targeting green R&D will help it gain critical mass to catch up with other R&D – a case of mitigating the adverse effects of path dependency. However, even when tax incentives are justified in principle, designing them so that they are cost-effective remains challenging. For example, how to ensure that incentives result in additional R&D instead of mostly subsidising activity that would have been undertaken anyway, and how to avoid relabeling of activities so that they become eligible for R&D tax incentives?

The present report focusses on environmentally related taxes, and less on emissions-trading systems as an alternative form of market-based instruments. Emissions-trading systems are very different from taxes in cases where tradable permits are allocated for free instead of being auctioned, as in those cases no public revenue is raised. Since public revenue is scarce, auctioning of permits in principle is preferred. In addition, permit allocation rules can affect market entry and exit decisions in ways that slow down the reduction of pollution, for example in cases where incumbent, pollution-intensive firms receive disproportionate allocations of permits. Also, for trading to work well, markets need to be well-designed and the number of trading parties large, a set of conditions that it is not straightforward to meet. These potential downsides of trading systems have to be weighed against the practice of environmentally related taxes (not the theory), in which it is not unusual to have preferential tax rates that blunt the environmental effectiveness of environmentally related taxes (see, *e.g.*, the discussion in Smith, 2008).

Emissions trading systems may perform better than taxes in cases where attaining a particular level of pollution is essential. This is because trading systems first define the level pollution abatement (or the cap on pollution) and then use the permit-trading mechanism to allow reaching the cap in a cost-effective manner. The permit price will only become known once trading takes place. With taxes, the price of every unit of pollution is known in advance. However, it is uncertain exactly how much abatement will take place, as many factors other than the tax influence that outcome. Nevertheless, knowing the price of pollution helps investors make abatement decisions, so taxes can give stronger abatement incentives than emission-trading systems.

### 3.2 *Raise revenues and use them in a socially productive way*

The thrust of the argument of section 3.1 is that market-based instruments in general are among the more cost-effective environment policy instruments, and should form part of an environment policy package that aims to deliver environmental improvements at the lowest cost. Market-based instruments also interact with broader tax and fiscal policy, not least because they raise revenue or at least have the potential to do so. As already mentioned, these revenues can be used in several ways, many of which are socially beneficial. This is a strong advantage of market-based instruments, but the risk exists that revenues would not be used well, in which case the appeal of market-based instruments declines considerably.

This section argues that strict earmarking of the proceeds from environmentally related taxes is not a straightforward choice within well-designed environmental tax policy, that environmental tax shifts are one potentially attractive way of recycling revenues, and that they potentially result in a double dividend.

#### *Earmarking revenues*

Revenues from environmentally related taxes, perhaps more often than other taxes, are subject to multiple and specific claims on their use, in particular when instruments are newly introduced. For example, it is often suggested that revenues from carbon pricing should be used for investment in low-carbon infrastructure, to reach international goals for climate finance, to compensate households or firms that are particularly strongly affected by carbon prices, to create green innovation funds, to fund carbon dividends, etc. These forms of revenues can make sense, but in general, decisions on what to fund should not guide how environmentally related taxes are designed or at which level they are set, and overly strict constraints on how to use revenues risk resulting in inefficient public spending.

It is sometimes argued that earmarking can help create support for environmentally related taxes. One reason is that earmarking can increase policy transparency, which is well-liked by voters. A second reason is that linking environmentally related taxes to environmentally related spending tends to increase support among part of the constituency. However, this can become counterproductive in the long run, as the need for flexibility in spending decisions rises and as environmentally related taxes become a standard feature of tax policy, which requires the broadest possible support. The higher the revenues from environmentally related taxes, the stronger the need becomes to ensure whole-of-government support for them. Statements of policy intent on how to use revenues can be a useful short-term way of facilitating the introduction of environmentally related taxes, which avoid the risks associated with strong earmarking.

In contrast to taxes, fees and charges are typically paid in proportion to an *ad quantum* of a service received, with revenues directed at specific beneficiaries (OECD, n.d.) – a form of earmarking. For example, taxes and charges are often levied by subnational and local governments, directed at public or private utilities for the provision of waste or water services. In principle, charges are thus different from taxes, in that their levels and design is based on recovering service costs.<sup>5</sup> In practice, the distinction between taxes and charges can be blurred, as many taxes include elements of cost-recovery. For example, fuel tax revenue in the United States is earmarked for the Highway Trust Fund (HTF), to finance investment in transport infrastructure, and some vehicle taxes are used to fund maintenance of transport infrastructure (Van Dender and Parry, forthcoming). Confounding of taxes and charges arises most often where rates vary by the source of pollution (e.g. charges for wastewater treatment that vary by polluter) or strongly reflect resource scarcity (e.g. charges on water abstraction that vary by source), but revenues are

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<sup>5</sup> Tax revenues tend to flow into the budget of national or regional governments, with no direct service received in exchange, thus including a certain redistributive element. Under the Pigouvian principle, taxes attempt to capture the cost of an additional unit of pollution to society, i.e. the marginal social cost.

still just or not sufficient to cover the service cost (Hogg et al., 2016).<sup>6</sup>

### *Recycling revenues and the potential for a double dividend*

In many countries, environmentally related tax revenue is used to reform the tax system to make it less distortive and more growth- and employment-friendly. Many OECD countries consider that shifting away from labour and corporate income taxes towards consumption and property taxes, would reduce the drag of tax systems on, or increase their support for, economic growth. Environmental fiscal reform, with its focus on the taxation of specific types of behaviour, can fit in this tax-shift model.

Significant environmental tax reforms were undertaken in some Northern European countries (e.g. Denmark, Finland, Norway, the UK, Germany and the Netherlands), but reform strategies and results differ strongly across countries. Most have introduced new environmentally related taxes, mainly levied on the energy consumption and CO<sub>2</sub> emissions, or revised existing ones, coupled with reducing corporate or personal income taxes, social security contributions or raising tax-free allowances (COMETR, 2007). More recently, such tax shifts were implemented in Slovenia (2002), Portugal and Ireland (both 2014). In the latter two, the tax shifts were part of broader efforts to increase fiscal sustainability in the context of the EU economic adjustment programmes implemented in these countries.

The environmental tax debate has benefited from detailed study of the efficiency- and growth-enhancing potential of environmental tax shifts, in the “double dividend” debate first proposed by Pearce (1991). Here, the first dividend of environmental tax reform would be the improvement of environmental quality. The second dividend is that cutting pre-existing and more distortionary taxes (offset with new environmental tax revenue) would result in a reduction of the economic efficiency-cost incurred for raising a given amount of tax revenue.

Goulder (1994) distinguishes between weak and strong versions of the argument. The weak double dividend claim is that returning tax revenues through cuts in distortionary taxes leads to cost savings relative to the case where revenues are returned lump sum, whereas the strong version holds that revenue-neutral swaps of environmentally related taxes for other distortionary taxes (e.g. direct taxes such as social security contributions or other taxes on personal or company income) would involve zero or negative gross costs. In relation to the strong double dividend, Bovenberg and de Mooij (1994) emphasise that improvements in social welfare depend on the specific properties of the distortionary tax that is modified.

Attempts to assess the empirical validity of the double dividend claim focus on carbon and other energy taxes and their effects on growth and employment, though sometimes investment is considered too (Hogg et al. 2016). Other types of environmentally related taxes are rarely discussed, an exception being a simulation of the impacts of different waste taxes on EU employment and output (Cambridge Econometrics, 2013)<sup>7</sup>. For carbon and energy taxes, the existence of a double dividend is usually analysed using computable general equilibrium or macro-econometric models, employing various assumptions on the type of revenue recycling chosen (Vivid Economics, 2012). These simulations indicate relatively consistently that some gains in employment are likely when increases in carbon or energy taxes are coupled with reductions in other taxes.

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<sup>6</sup> More detailed discussions of experience with earmarked taxes in the road transport sector can be found in Parry and Van Dender (forthcoming). Borenstein (2016) discusses the economics of fixed-cost recovery of utilities with an application to the electricity sector, but most of the insights regarding revenue use apply more broadly.

<sup>7</sup> This analysis finds that if the revenues of landfill and incineration taxes are used to decrease employers' labour cost – taxes on waste have a negligible impact on employment and GDP.

A recent study for 27 EU countries (EU 28 except Croatia; Groothuis, 2016) uses the macroeconomic E3ME model to investigate the effect on growth and employment of shifting taxes from labour towards fossil fuels and carbon emissions, while also increasing standard and reduced VAT rates and raising taxes on electricity and water use for large users. A total of EUR 554bn is shifted from labour towards natural resources and consumption, which is equivalent to 13% of labour tax revenue and results in a 5.6%-point reduction of average personal income tax rates. The econometric model suggests that gradually introducing this tax shift over the period 2016 – 2020 would increase employment by 3% in 2020, GDP would rise by 2%, while water use, energy use and carbon emissions would decline by more than 5%. This means that there would be a substantial double dividend.

Overall, theoretical analysis and *ex ante* evaluations suggest that the potential for a double dividend exists, but that it should not be taken for granted that *any* shift from business or labour income taxes towards environmentally related taxes results in a less distortive tax system. The reason for guarded optimism is that environmentally related taxes can also be a relatively inefficient way to raise revenue, given their narrow tax base compared with income taxes (requiring high rates to raise a given amount of revenue). Much depends on precisely what the pre-reform tax system looks like, and what precisely will be done with the revenues to alleviate pre-existing distortions (instead of making them worse).

Realising the growth-enhancing potential of environmental tax reform requires customised analysis and context-specific reforms. However, real-world tax systems likely offer plenty of options for reform that would be efficiency improving, and could also make the tax system more progressive.<sup>8</sup> Simultaneously, uncertainty about the precise nature of the double dividend reinforces the need to emphasise the environmental effectiveness of policy instruments in their design. This includes safeguarding against unproductive uses of revenues; in particular precisely in ways that compromise on the environmental effectiveness of environmentally related taxes, as is discussed next.

#### *Implicit versus explicit revenue use*

This report distinguishes between “implicit” and “explicit” uses of tax revenues. Explicit revenue use refers to the deployment of those revenues that actually flow into public coffers, which can be used to cut other taxes (keeping total tax revenue more or less constant), cut debt, to increase government investment or expenditure, to fund transfers, etc. All of these can be useful, dependent on a country’s situation at a particular point in time, and have been used in a range of countries around the world, as described in the previous paragraphs.

Implicit revenue use, in contrast, refers to the deployment of revenues that are not actually raised, e.g. because of preferential environmental tax rates for some users or uses, or because of free allocation of permits in emissions-trading systems. In other words, implicit revenue use refers to the deployment of foregone revenue, where the benchmark for establishing foregone revenue is a uniform rate in the case of environmentally related taxes and full auctioning in the cases of emissions trading systems. Implicit revenue use consequently is the provision of compensation to some polluters, and providing this compensation via a modification of the environmental pricing instrument itself (lower rates or free permits).

It is well understood that in the case of taxation, such compensation results in reduced effectiveness of the environmental policy, and that this often can and should be avoided by providing compensation in different ways (lower corporate income tax rates, for example), in as far as compensation is justifiable (see section 3.3). Providing compensation through free emissions permits maintains the price signal at the margin, and is in that sense less detrimental to environmental incentives. However, providing

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<sup>8</sup> Brys et al. (2016) discuss tax policy reforms that could improve efficiency and equity of the tax system.



permits for free does modify average permit rates, and this can affect the ranking of investment projects. Depending on the specifics of permit allocation rules, there is a risk that more environmentally ambitious investment projects move down the ranking because they would result in fewer free permits obtained. This implies that moving towards full auctioning of permits will improve the abatement signals from emissions trading.

### 3.3 *Protecting the competitiveness of domestic industries*

A prominent concern is that pricing environmental externalities may adversely affect the ability of domestic groups (e.g. specific industries or firms) to compete, although the available empirical evidence reveals little to no effects. The real or perceived possibility of environmentally related taxes to affect the competitiveness of domestic industry has an effect on the design of environmentally related taxes in all of the policy areas surveyed in this paper.

In relation to carbon and energy taxes, but also carbon trading systems, there is growing econometric evidence that the immediate competitiveness impacts of existing carbon pricing mechanisms are negligible or nil (Arlinghaus, 2015; Partnership for Market Readiness, 2015). While this partly can be explained by the low prices and free allocation prevailing in most mechanisms, these same prices have reduced emissions, and windfall profits have occurred, so it is not the case that prices have always been ineffective environmentally or trivial economically.

Stringent environmental policy does not necessarily impinge upon strong firms' capacity to realise productivity growth. Albrizio et al. (2014) find that enhanced environmental policy stringency is associated with a subsequent increase in short-run productivity growth of the most productive industries and firms. In the longer run, substitution possibilities are larger, and stronger firms will be able to exploit them, potentially resulting in improved competitiveness. Where flanking measures to alleviate competitiveness concerns are used, support to industry is often not well targeted to address relocation risk (i.e. carbon leakage). For example, over the current trading period of the EU ETS, 43 percent of allowances is freely allocated to industry deemed at risk of carbon leakage, but the criteria used to distribute allowances to firms are not always very closely related to actual relocation or downsizing risk (Martin et al., 2014). In particular, trade intensity, which implies a free allocation to 75% of the subsectors in the EU ETS, ignores that most of the trade exposed firms have a very low-carbon footprint and are therefore immune to a carbon price.

Beyond carbon and energy taxes, evidence on the effects of environmentally related taxes on the competitiveness of domestic industries or production is much scarcer. Ecotec (2001) surveys the design of nine types of environmentally related taxes and charges (NO<sub>x</sub>, water abstraction, waste water discharges, pesticides, fertiliser, landfill, aggregates, packaging and batteries) in EU member states, finding that the competitive position of affected sectors is the major concern expressed in relation to the design of these taxes and charges. As a result, the largest polluters are frequently exempt from the instrument. While this may be the reason for the absence of any effects of these instruments on the competitive position of the polluting industries, these exemptions or reduced rates are also found to severely compromise on their environmental effectiveness.

Ecotec (2001) also finds that exemptions are too often granted based on a static assessment of the effects of market-based policies, ignoring their dynamic efficiency effects. These arguments relate closely to the discussion of the design of carbon and energy taxes. However, some of environmental goods and services affected by the instruments surveyed in Ecotec (2001) are less widely traded (e.g. waste) which may suggest that their effects are more likely to unfold domestically than in the case of carbon or energy taxes, decreasing the relevance of arguments around leakage.

### 3.4 *Avoiding adverse effects on the distribution of income*

Concerns about the distribution of income and the affordability of goods or services strongly shape the design of taxes on energy products and electricity and water abstraction charges. In the case of energy products, taxes are generally too low to reflect the costs of pollution or climate change (e.g. OECD, 2016). Water prices barely recover the cost of water provision and much less the increasing scarcity of the resource (see also section 4.4).

Price increases translate into higher direct costs for households and put upward pressure on other prices due to their flow-on effects. This may pose affordability problems for lower-income households, depending on their specific income and expenditure profiles. Targeted flanking policies for the most vulnerable households, designed such that they do not compromise the environmental effectiveness of a price instrument e.g. using income-tested compensation or lump sum transfers, can help mitigate such adverse affordability impacts. This, however, presupposes knowledge of the precise distributional impacts of price increases.

Flues and Thomas (2015) examine the distributional effects of energy taxes for 21 OECD (mainly European) countries using household expenditure microdata, i.e. expenditure measured at the household level.<sup>9</sup> The analysis finds that taxes on transport fuels are roughly proportional on an income basis and tend to be progressive on an expenditure basis. In contrast, taxes on heating fuels are found to be slightly regressive, and taxes on electricity are more regressive on both an income and expenditure basis than taxes on heating fuels. Flues and Van Dender (2017) use similar household-level spending data to investigate the affordability of heating fuels and electricity, at current prices and in a counterfactual energy tax scenario in 20 OECD countries. A combination of an energy tax increase with income-tested compensation can improve energy affordability for the poorest population groups using just a third of the additional revenues raised. Consequently, even after having implemented effective flanking policies, substantial amounts of revenues remain available for other uses.

In the case of water, the analysis of distributional effects and affordability concerns rarely focusses on abstraction charges or taxes on wastewater treatment specifically, but on household income or expenditure shares on water and sanitation bills as a whole. OECD (2010) shows that water and sanitation bills do not usually pose a considerable burden on households on average (among the countries analysed, disposable income shares range from 0.2% to 1.2%), but the share of disposable income spent on water supply and sanitation bills more than doubles (to 1% to 7.9%) when just the poorest population decile is considered. The design of water prices is, however already strongly shaped by concerns about affordability. The low average prices rarely reflect the cost of service provision and, tariff structures often feature social rates.

In contrast to the debate about the affordability of energy and fuel poverty, indicators for water affordability are much less commonly accepted (Mahmood and Sharma, 2009; see also Flues and Van Dender, 2017, for a discussion of the indicators to measure energy affordability). Governments and international organisations report indicative water affordability benchmarks (threshold expenditure shares), for example, 3-5% by the World Bank, 3% by the UK, and 2.5% by the United States (ibid.). Comparing these shares to the calculations reported in OECD (2010) suggests that there are some OECD countries in which water affordability may pose a problem for the poorest population groups.

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<sup>9</sup> Distributional effects can be assessed by measuring tax burdens as a percentage of income across the income distribution and as a percentage of expenditure across the expenditure distribution. Although there are arguments for using either measure, a key issue is that households with low transitory income, who are not lifetime poor, strongly affect average income while they do not affect average expenditure as much. The analysis in Flues and Thomas (2015) emphasises the analysis on an expenditure basis, but shows income-based results too.

The economic literature notes that the practice of keeping water prices artificially low across all expenditure deciles often leads to deteriorating infrastructure and services, which may disproportionately affect the poorest households as resorting to alternative options such as bottled water is more expensive. The literature also notes that cost-reflective design of water prices, including scarcity-based water pricing using abstraction charges, and taxes or charges for sewage and wastewater treatment, does not have to be at odds with decreases in water affordability for vulnerable groups. Carefully designed tariff structures, but also non-tariff instruments (e.g. income support) can be used to compensate vulnerable groups for higher, cost-reflective water prices (e.g. Rogers et al., 2002; OECD, 2010).

#### **4. The state of environmental fiscal reform**

The previous section argued that environmentally related taxes are among the most cost-effective environment policy instruments and that judicious use of the revenues that they raise adds further benefits. In practice, environmentally related taxes have been introduced to steer towards less polluting forms of behaviour as well as to raise revenue, and in some cases they are shaped by the “user charge” idea — i.e. they are seen as charges that help fund infrastructure and operation costs in particular sectors. User charges often are more akin to average cost pricing, so that environmentally related tax practice becomes a hybrid of average and marginal cost pricing. Tax practice also is shaped by administrative considerations (where a trade-off can exist between environmental effectiveness and ease of administration), by equity and poverty-related considerations, by potential effects on firms’, sectors’, and countries’ ability to compete, and by interest groups in general. The weights of these different factors differ between the main sectors in which environmentally related taxes are used and across countries, resulting in different profiles of environmentally related taxes across the economy and among countries.

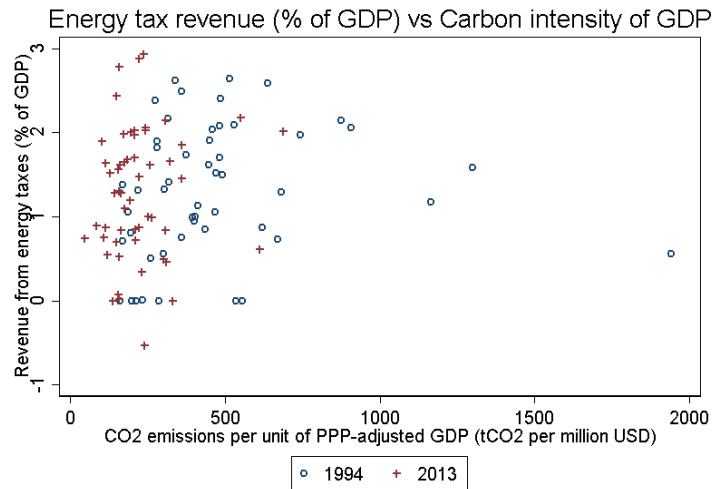
Section 4.1 discusses how much revenues are currently raised from environmentally related taxes, Sections 4.2 through 4.7 consider the practice of environmentally related taxation in the following areas: energy, transport, waste, water, and chemicals, by way of their practical relevance in terms of the scope of their usage and the revenues they raise.

##### **4.1 Revenues from environmentally related taxes**

This section discusses the revenues raised through environmentally related taxes and how they compare to countries’ GDP. High revenues from environmentally related taxes are often taken as a sign of success, in the sense of reflecting strong political effort to reduce environmentally harmful behaviours using taxes. However, they could also be an indicator of weakness in the sense that high revenues indicate high remaining pollution, and they do not reflect the possibility that countries use different policy instruments or rely heavily on tax incentives to pursue environment objectives. Here, like in most analyses, the emphasis is on the first interpretation, i.e. taking higher revenue shares as mainly indicative of stronger effort, particularly effort based on more cost-effective policy instruments.

A large econometric literature shows that taxing the carbon content of energy use leads to lower energy consumption in the long run (for just one recent example, see Sen and Vollebergh, 2016). However, because the responsiveness of energy consumption to tax rates is not fully proportional, revenues can remain high when consumption is reduced even if tax rates do not rise to balance the negative revenue effect from the shrinking tax base. This means that energy taxes can both be effective environmental and effective revenue raising instruments. Figure 4 indicates that there is no strong positive or negative correlation between an economy’s carbon intensity and revenues from energy taxes in 50 OECD and partner economies. The variation is mostly cross-sectional, with large inter-country differences relating to economic and policy structures. Also, between the two years shown, other factors than energy taxes affect the size of the base, including income growth and pre-tax energy prices. Absence of a negative correlation therefore does not indicate that taxes are not strongly used, just like absence of a positive correlation does not support the view of limited usage.

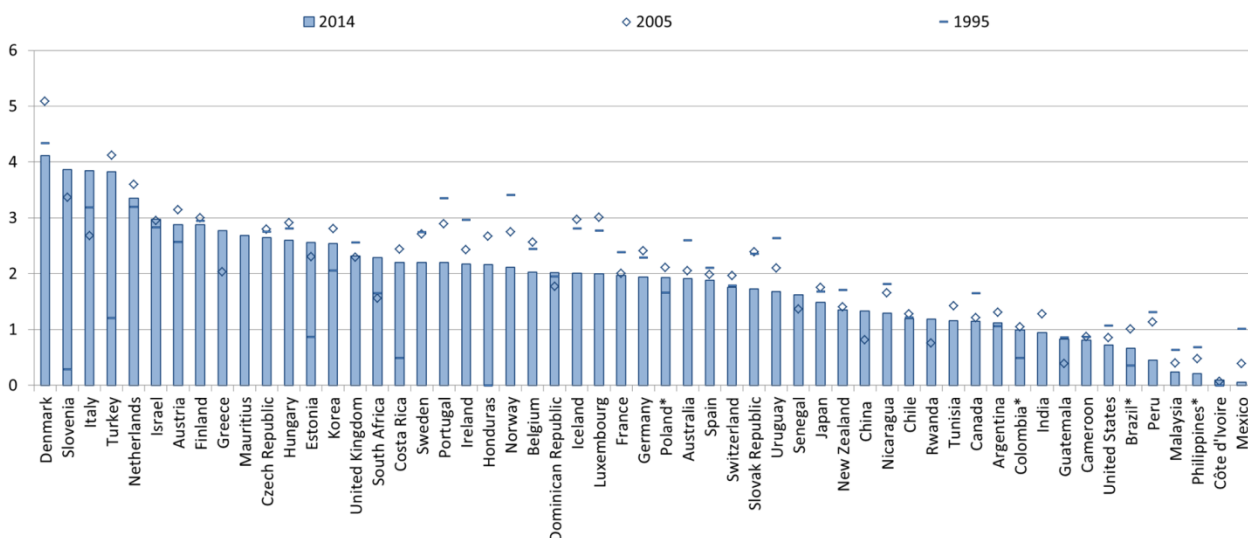
**Figure 4 Revenue from energy taxes compared to GDP and carbon intensity of GDP OECD and selected non-OECD economies, 1994 and 2013**



Sources: Own calculations based on OECD *Database on instruments used for environmental policy* (<http://stats.oecd.org/>) and World Bank *World Development Indicators* (<http://data.worldbank.org/>).

Environmentally related tax revenue varies across OECD and other economies when compared to GDP (Figure 5), ranging from 0.06% of GDP in Mexico to 4.1% of GDP in Denmark in 2014. The average share in 2014 is at 1.8% and the median is 2%. Considering the trend over time reveals a downward tendency of revenue shares since 1995 in most countries. This usually is attributed to the non-increase of nominal excise rates and the high petrol prices that negatively affected environmentally related tax bases in the recent past. In some countries, a shift towards less taxed diesel also reduces revenues. However, time patterns differ among countries, with a few countries exhibiting raising revenue shares since 1995 and other countries characterised by more stable revenue-to-GDP ratios.

**Figure 5 Environmentally related tax revenue compared to GDP in OECD and selected non-OECD economies, in 2014, 2005 and 1995**

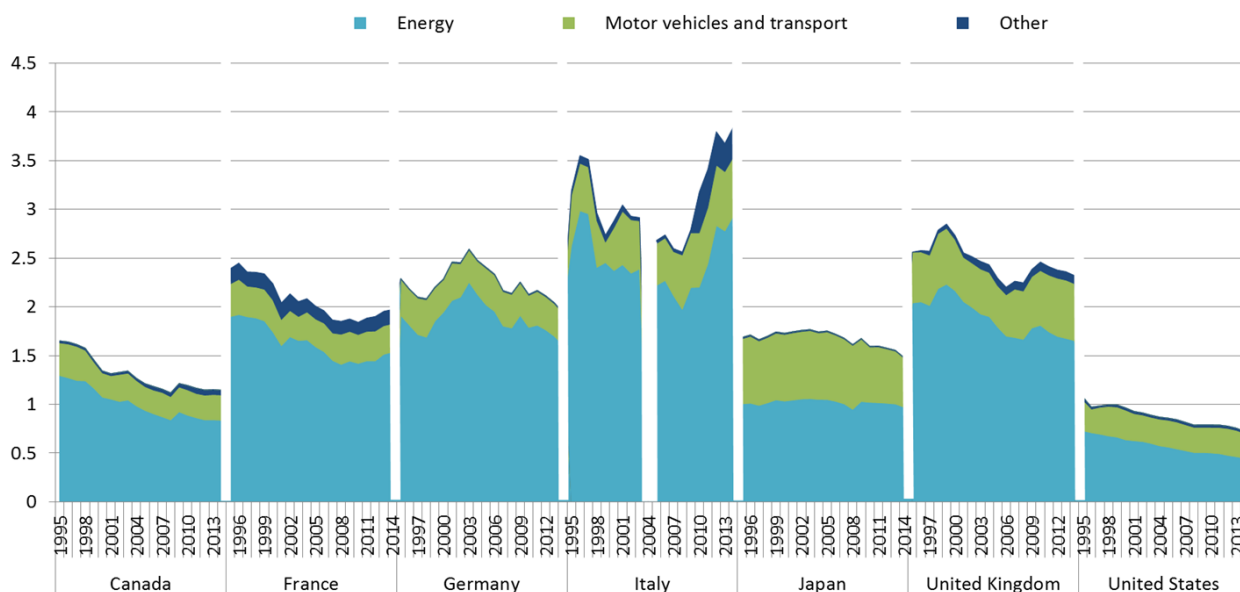


Note: \* indicates that data are for 2013 instead of 2014

Source: OECD *Database on instruments used for environmental policy*, <http://stats.oecd.org/>.

These varying patterns of revenue shares over time are also present in G7 economies. Figure 6 presents the trend of environmentally related tax revenue compared to GDP for the G7 countries between 1995 and 2014. Revenue shares demonstrate an overall downward trend in most G7 countries. However, in Italy, the revenue share is higher in 2014 than in 1995 and in France the revenue share is again increasing in recent years.

**Figure 6 Environmentally related tax revenue compared to GDP in G7 economies by component: 1995 - 2014**



Note: 2004 data for Italy are missing.

Source: OECD Database on instruments used for environmental policy (<http://stats.oecd.org>).

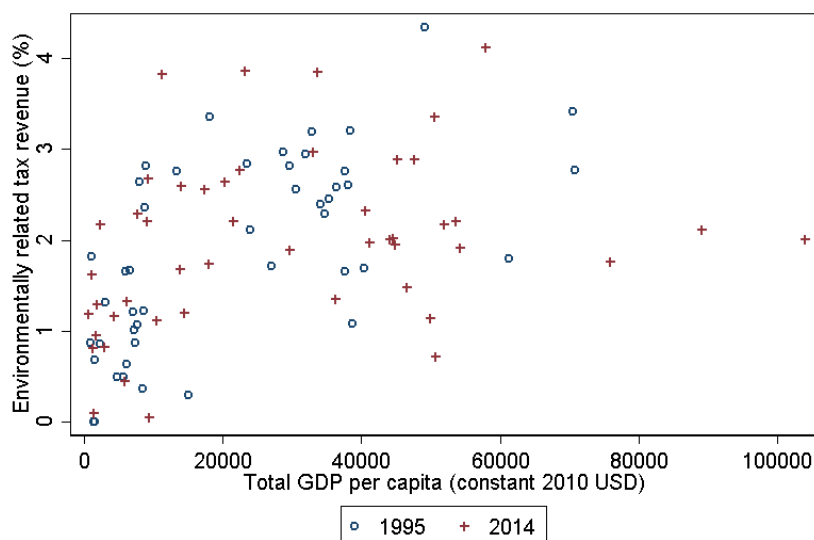
Figure 6 also shows the composition of environmentally related tax revenue, where the pattern observed in the G7 countries is very similar to the patterns observed in other OECD and other G20 economies (i.e. the other countries included in the OECD database). By far the largest share of revenues raised is from taxes on energy. Within taxes on energy use, as will be seen below, most are levied on transport fuels. Taxes on motor vehicles and other transport taxes raise much less revenue than energy taxes, but the revenue shares can still be considerable. Other environmentally related taxes raise much less revenue. In as far as improved environmental tax practice would result in similar structures of revenue (which arguably is the case, see below), the focus of discussion of environmental taxation as an element of broader tax policy is and will remain on energy and transport taxes.

The dominance of transport-related taxes in the revenues included in the figures, and transport fuel taxes in particular, is one good reason to refer to “environmentally related taxes” rather than “environmental taxes”. Transport fuel taxes clearly are relevant to the environmental impact of transport activities, but they have not historically been introduced for that reason. The focus on revenue-raising, rather than mitigating negative environmental impacts, helps explain why transport-related taxes are not always well aligned with environmental impacts from energy use. One example of misalignment is the existing tax differential between petrol and diesel taxes in most countries. Although burning diesel emits higher levels of carbon dioxide per litre than gasoline and, depending on the technology employed, often also more harmful air pollutants, diesel is taxed at a lower rate than petrol in most OECD and G20 countries, and beyond (Harding, 2014). Despite relatively high energy taxes in transport (even if lower than external costs in many instances), compared to other sectors, and despite their clear impact on fuel use in the sector, transport is still very heavily dependent on fossil fuels, and the revenue raised from this base

is not negligible from a tax policy perspective.

Figure 7 plots environmentally related tax revenue compared to GDP (vertical axis) against GDP per capita (vertical axis), in 1995 and 2014, for the same set of countries as included in Figure 6. The data do not suggest a particularly strong correlation between the revenue share and per capita GDP, either in a cross-sectional sense (i.e. looking at one year in isolation) or comparing points in time, with perhaps the exception of the lowest incomes (roughly, less than USD 10,000) that tend to go together with comparatively low revenue shares.

**Figure 7 Environmentally related tax revenue compared to GDP and GDP per capita, 1995 and 2014**



Sources: Own calculations based on OECD *Database on instruments used for environmental policy* (<http://stats.oecd.org/>) and World Bank *World Development Indicators* (<http://data.worldbank.org>).

What is the revenue potential from environmental taxation? This is a hard question to answer, as estimates depend on what is understood under good or optimal environmental tax practice, and quantification becomes increasingly uncertain as more ambitious reforms – for which behavioural adaptations are harder to estimate – are considered.

The revenue impacts of a move towards marginal-social-cost pricing, starting from the current situation, are considered in a report on the EU-28 countries.<sup>10</sup> Such a reform would involve: applying a uniform carbon and energy tax component on energy use within sectors; increased vehicle taxes; greater use of ticket and tonnage taxes in aviation transport; raising landfill and incineration taxes; focusing packaging taxes more on prevention and not just recycling; taxing plastic bags; increasing taxes on air and water pollution; applying more systematic water abstraction charges (per volume) and effluent charges (dependent on biochemical oxygen demand); integrating more pollution-based differentiation into water charges; taxing pesticides more on their potential environmental impact than on their active ingredients; and taxing fertilizers more broadly.

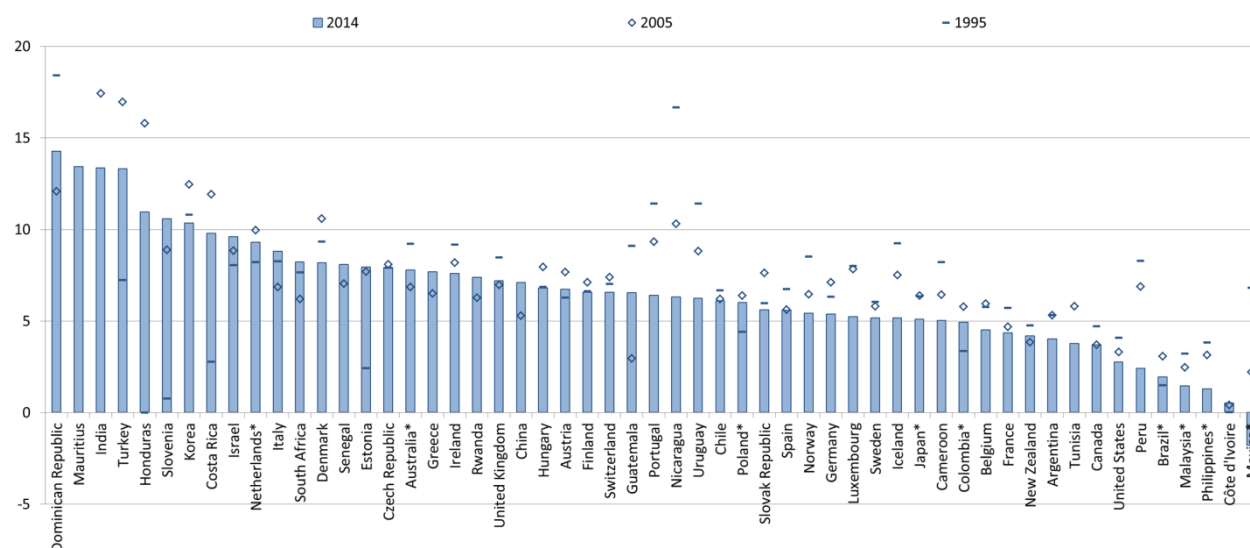
Across the EU-28, the report estimates that the current revenue from environmentally related taxes amounts to 2.6% of GDP. Moving to the improved practice described above would increase this share by slightly more than a percentage-point, to 3.6% of GDP. Most of this increase would come from increased revenue from transport fuels and – notably – from taxes on transport vehicles (equal to a half

<sup>10</sup> Eunomia et al. (2016), “Study on assessing the environmental fiscal reform potential for the EU-28”, EC, Brussels

percent of GDP). Tax revenue from non-transport related sources would contribute only an extra 0.2% compared to GDP.

Besides marginal cost pricing, another reference point related to environmentally related tax revenues deserves mention. In the *Roadmap to a Resource Efficient Europe*, the European Commission calls for a shift away from taxing labour towards environmentally harmful activities by 2020, in line with the best practice of European member states.<sup>11</sup> A 10% share (EU average) of environmental taxes in total tax revenues would be in line with the best performing member states in 2009.<sup>12</sup> In 2014, the share of environmentally related tax in total tax revenue averages at 6.5% across all countries included in Figure 4 and at 5.3% across the G7 countries. Similar to the share in GDP above, the share in total tax revenue falls in most countries over time. Only few countries have brought up environmentally related tax revenue as a share of total tax revenue since 1995. Attaining the 10% share hence would imply considerably higher environmentally related tax rates, and a reversal of the trend observed in recent years.

**Figure 8 Environmentally related tax revenue as a share of total tax revenue in OECD and selected non-OECD economies, in 2014, 2005 and 1995**



Note: \* indicates that data are for 2013 instead of 2014

Source: Own calculations based on OECD *Database on instruments used for environmental policy* (<http://stats.oecd.org/>).

#### 4.2 Taxing energy use

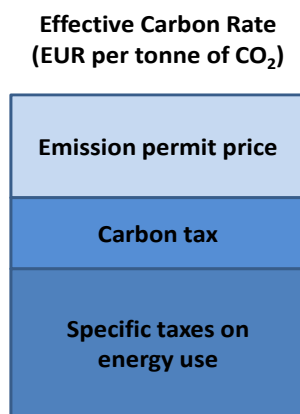
Taxes on energy use are the main source of environmentally related tax revenue. But what is their structure across sectors, and how well are they aligned with what the logic of environmental taxation would suggest? This section provides some insights, drawing from the OECD's report on Effective Carbon Rates (OECD, 2016a). OECD (2016a) is a comprehensive stocktaking of taxes on energy use and prices of tradable emission permits that together constitute a carbon price, in 41 OECD and G20 countries. Together, these countries represent 80% of global energy use and of the CO<sub>2</sub> emissions resulting from it. The Effective Carbon Rates include explicit carbon taxes and price signals from emissions trading systems but also excise rates (Figure 5). Excise taxes on energy are included in the definition of effective carbon

<sup>11</sup> Communication from the Commission to the European Parliament, the Council, the European Economic and Social Committee and the Committee of the Regions, *Roadmap to a Resource Efficient Europe*, COM(2011) 571 final.

<sup>12</sup> Commission Staff Working Paper, *Analysis associated with the Roadmap to a Resource Efficient Europe. Part II*, [SEC\(2011\) 1067 final](#).

rates because their behavioural impacts that are identical or at least strongly similar to those of explicit carbon pricing instruments, i.e. they are environmentally related taxes.<sup>13</sup>

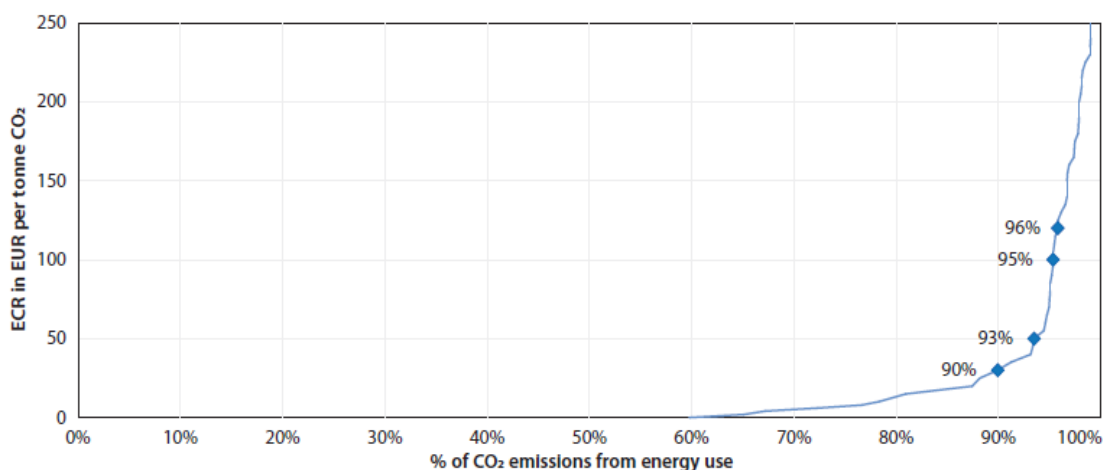
**Figure 4: Components of the effective carbon rate**



Source: OECD (2016a)

Figure 5 shows the distribution of effective carbon rates across all carbon emissions from energy use in the 41 countries included in the analysis. It shows that a large share of emissions is subject to a low effective carbon rate and a very small share is subject to high rates. More specifically, 60% of emissions from energy use are not priced, 90% are priced at less than EUR 10 per tonne of CO<sub>2</sub>, 93% at less than EUR 50 per tonne of CO<sub>2</sub>, and 96% at less than EUR 120 per tonne of CO<sub>2</sub>. Taking a conservative estimate of the social cost of carbon emissions, EUR 30 per tonne, this indicates a major gap between current practice and implementation of the polluter pays principle. The combined concerns of climate change, local air pollution, and scarcity of public revenue very strongly point towards the desirability of reducing this gap.

**Figure 5: Proportion of CO<sub>2</sub> emissions from energy use subject to different levels of effective carbon rates across 41 OECD and selected partner economies**



Source: OECD (2016a)

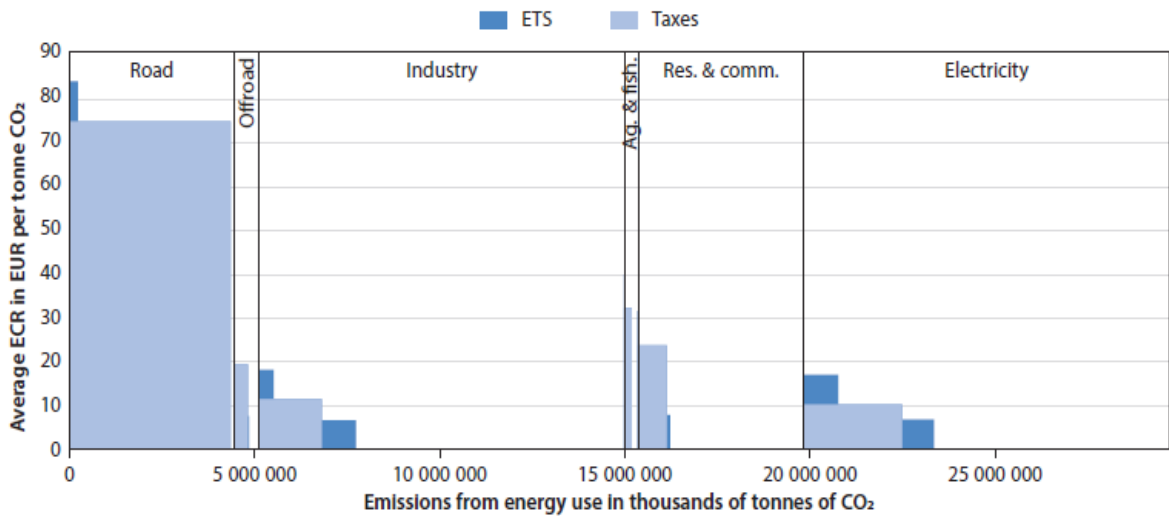
<sup>13</sup> There are reasons to think that the different components of the effective carbon rates can affect different behavioural margins differently, e.g. where investment incentives are concerned, or where new carbon taxes could be more salient than long-standing excise rates. This level of granularity is beyond the scope of the present analysis, however, and the effective carbon rates construct remains useful as a first approximation to the structure of the price on carbon resulting from the different specific pricing instruments.



Figure 6 shows the effective carbon rates for the 41 countries as a whole, for six economic sectors (road transport, off-road transport, heating and process use, agriculture and fishery, and electricity). It also breaks down the effective carbon rates into tax and ETS components. The following observations are worth noting:

- The effective carbon rates are much higher in road transport than in the other sectors of the economy. These high rates, combined with a relatively inelastic tax base, are the reason why the share of transport tax revenue dominates in total environmentally related tax revenue. To avoid confusion, it is mentioned here that “relatively high rates in transport” should not be interpreted as “sufficiently high rates in transport”, because whether rates are appropriate or not is not a matter of comparison between sectors but of comparing rates to marginal external costs. Marginal external costs differ between sectors, and are higher in transport, particularly when marginal external congestion costs are taken into account.
- In road transport, practically all emissions are covered by a price. This is not true of other sectors, where sometimes there is only a tax, or only a trading system, or both, or neither. In cases where both instruments apply, the question of policy interactions presents itself: if both instruments aim to cut carbon, it will be the cap of the trading system that determines the environmental outcome as long as it is binding. However, taxes can exist for other reasons (other market failures or revenue raising objectives), in which case the question is whether pricing policies are well-co-ordinated.
- The strength of the price signal from taxes, and in particular excise taxes on energy use, as opposed to the price signals from carbon taxes, dominates those of emissions trading systems. This is very strongly the case in the road transport sector, but remains true of other sectors where emissions trading is more prevalent. Regional or country breakdowns, focussing on jurisdictions where trading systems are in place, change this picture for industry and particularly for the electricity sector, but the strength of carbon price signals from trading systems are, on average across the emissions base, always weaker than those of taxes (see, e.g., Van Dender et al, 2016, for a discussion of taxation and emissions trading in the energy sector in the European Union).
- The limited contribution of trading systems to the effective carbon rates is due to both narrow coverage of emissions (across the entire emissions base) and to low permit prices. Nevertheless, higher permit prices would result in considerably higher effective carbon rates in the industry and in electricity sectors, in countries or regions where trading systems operate.

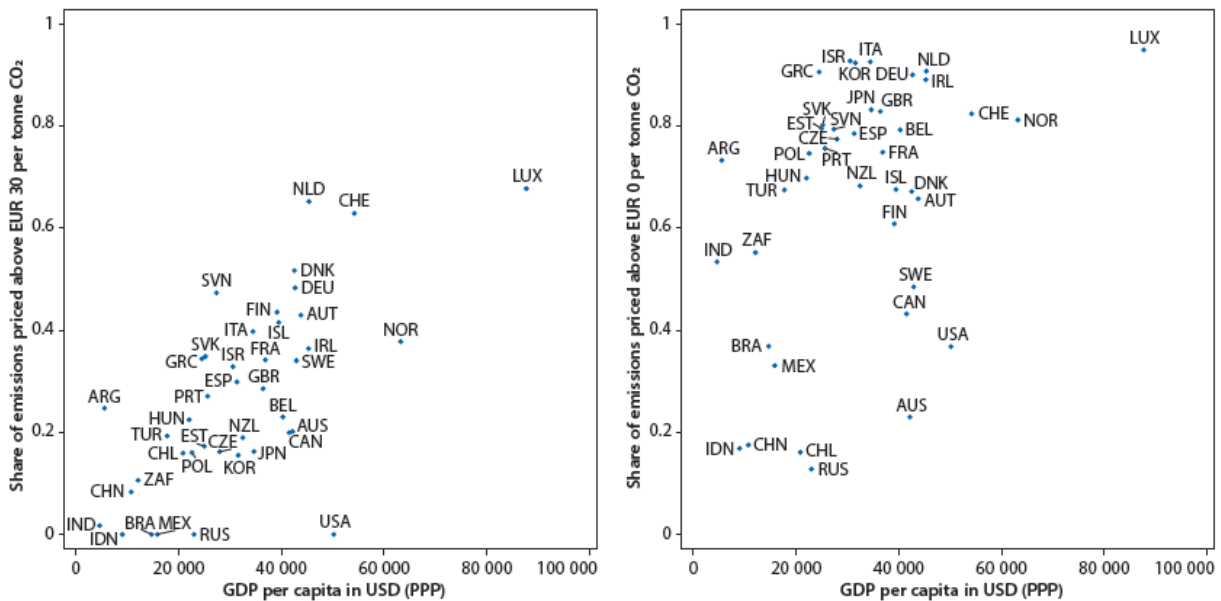
Figure 6: Average effective carbon rates across the 41 economies by sector and price instrument



Source: OECD (2016a)

A detailed analysis of effective carbon rates and their effects is well beyond the scope of this report, but the next three sets of figures provide some high-level indications. Figure 7 plots the share of emissions priced above EUR 30 per tonne of CO<sub>2</sub> (left) and above EUR 0 per tonne of CO<sub>2</sub> (right) against GDP per capita. The pattern suggests that more emissions are priced above EUR 30 as GDP per capita increases, and that there is a similar but weaker correlation when considering the share of emissions priced at some positive rate. Higher GDP correlates with higher rates, but as was seen before, not so much with higher revenue, indicating that over the long term the energy tax base is fairly elastic.

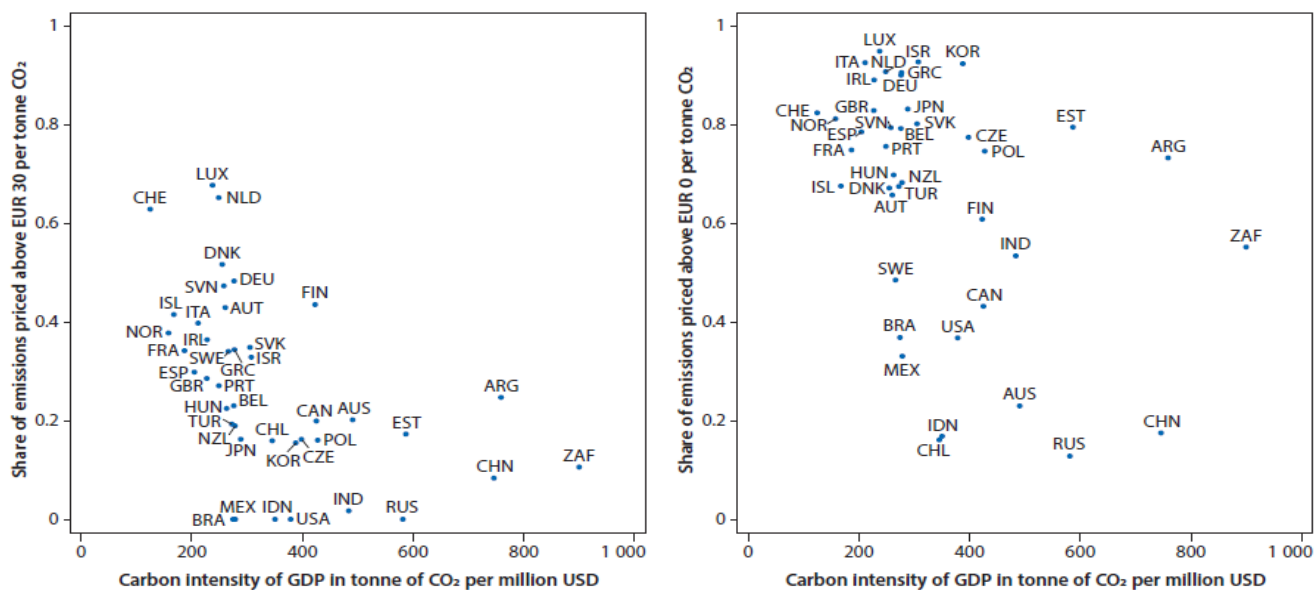
Figure 7: Proportion of CO<sub>2</sub> emissions priced above EUR 30 (left) and EUR 0 (right) per tonne of CO<sub>2</sub> relative to GDP per capita



Source: OECD (2016a)

Figure 8 plots the share of emissions priced above EUR 30 per tonne of CO<sub>2</sub> (left) and above EUR 0 per tonne of CO<sub>2</sub> (right) against the carbon intensity of GDP, suggesting a negative correlation in both cases (but again weaker in the right panel and in general showing more variance at lower price levels). Causality can run both ways, but it is plausible, and confirmed by microeconomic evidence, that higher prices also do contribute to lower carbon intensity of GDP.

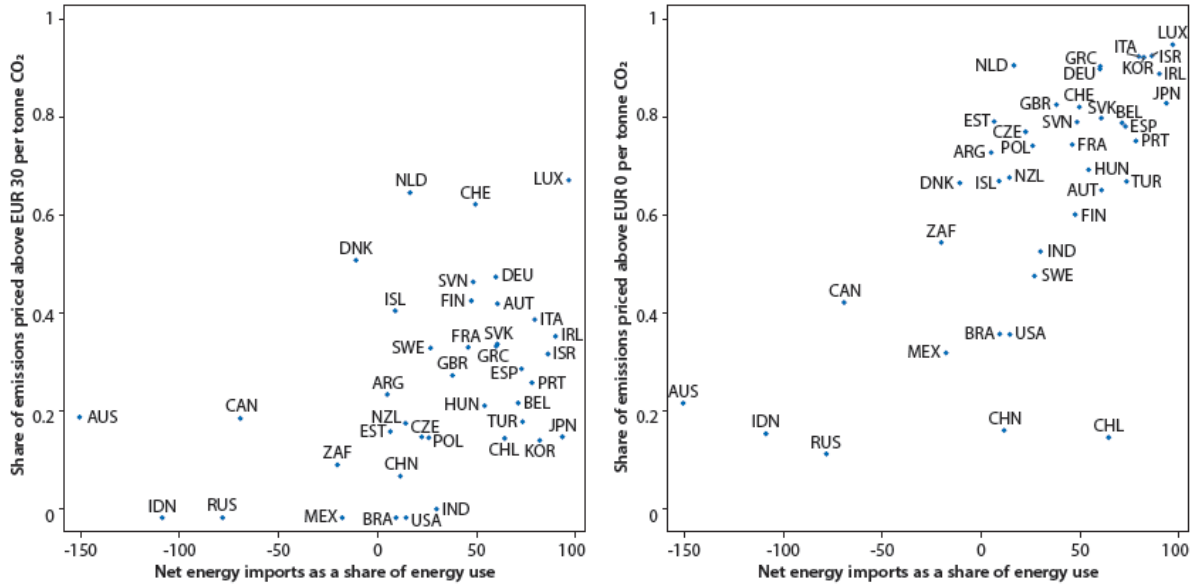
**Figure 8: Proportion of CO<sub>2</sub> emissions priced above EUR 30 (left) and EUR 0 (right) per tonne of CO<sub>2</sub> relative to the carbon intensity of GDP, 41 countries, 2012**



Source: OECD (2016a)

Figure 9 plots the share of emissions priced above EUR 30 per tonne of CO<sub>2</sub> (left) and above EUR 0 per tonne of CO<sub>2</sub> (right) against the net share of energy imported. Net exporters price energy less and the higher the share of net imports, the higher the share of emissions priced at high or positive rates. While there is in principle no immediate economic reason for such a pattern to prevail, political economy arguments may help explain the observed patterns.

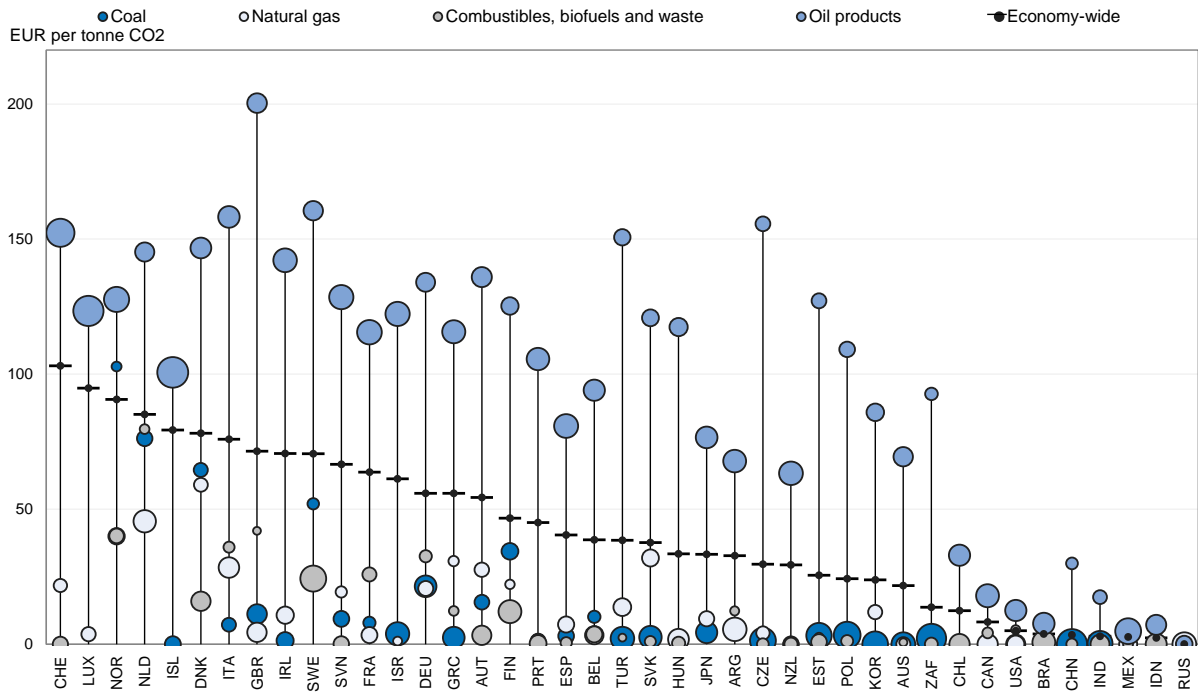
**Figure 9: Proportion of CO<sub>2</sub> emissions priced above EUR 30 (left) and EUR 0 (right) per tonne of CO<sub>2</sub> relative to net energy imports**



Source: OECD (2016a)

The discussion of energy pricing up to now has focussed on the role of taxes and trading systems to form carbon prices. The inclusion of trading systems makes it impossible, given data availability, to express the effective carbon rates by fuel. Figure 10 restricts attention to energy taxes, showing how they differ by fuel, for a selection of OECD and G20 countries. The most striking observation, of course, is that oil products are taxed more than other energy sources. This is because of their widespread use in transport (where rates are high) but not only – oil products used for non-transport purposes are also taxed more highly than other fuels in most cases. Coal tends to be taxed at very low rates or not at all across all OECD and G20 economies, despite its high carbon content and relatively unappealing air pollution profile in general. Given the low rates and coverage of carbon trading systems discussed above, their inclusion would be unlikely to change these general messages.

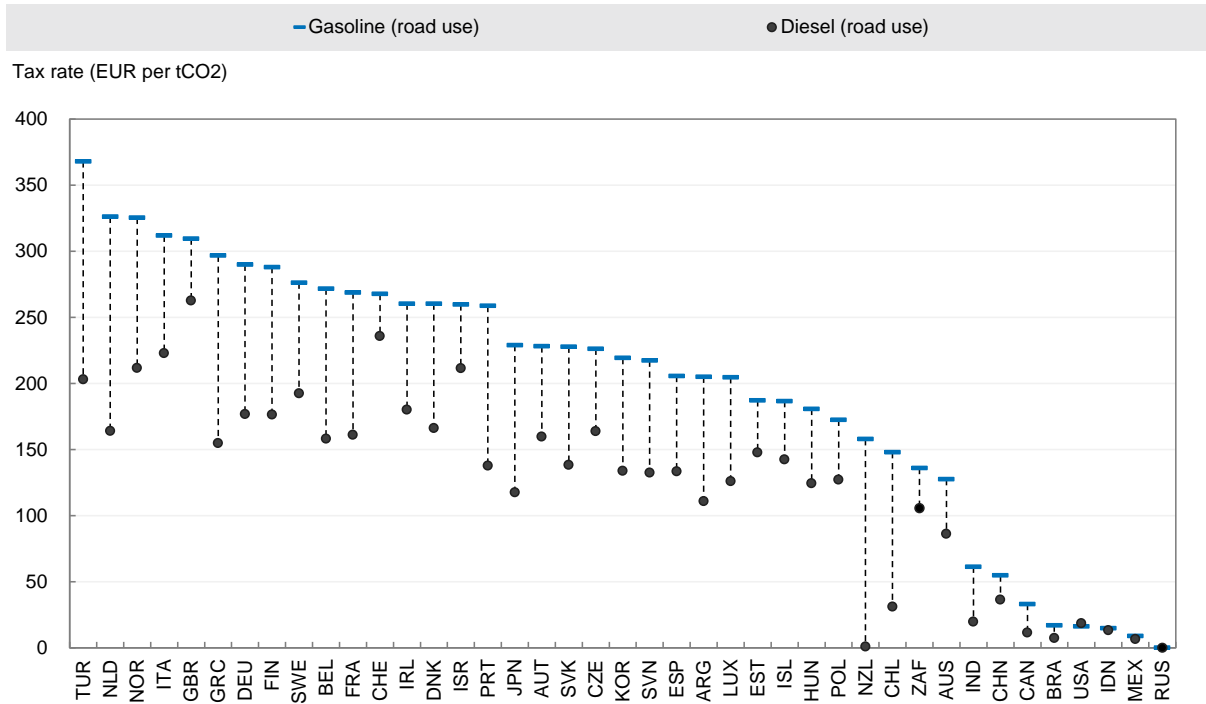
**Figure 10: Summary of carbon emissions from energy use and effective tax rates on carbon from energy use for each fuel and on an economy-wide basis**



Source: OECD (2015b), using data on carbon emissions from energy use for 2012, sourced from IEA (2014)

Figure 11 reveals another widespread feature of energy taxation by fuel type, namely that diesel for road use is taxed less (per GJ but also per tonne of CO<sub>2</sub> emitted, as shown in Figure 11) than gasoline. This is the opposite of what would be expected on environmental grounds. Diesel contains more carbon per litre than gasoline, and for most current pollution control technologies also emits more air pollutants. Finally, to the extent that diesel engines are more efficient and hence produce more distance per litre than gasoline vehicles, and to the extent that fuel taxes are intended to capture driving-related transport externalities, this too suggests that diesel taxes should exceed gasoline taxes on a per litre basis. Awareness of the adverse environmental structure of transport fuel taxes, combined perhaps with revenue considerations, is beginning to result in efforts to reduce or close the “diesel differential” in several OECD countries and within the G7.

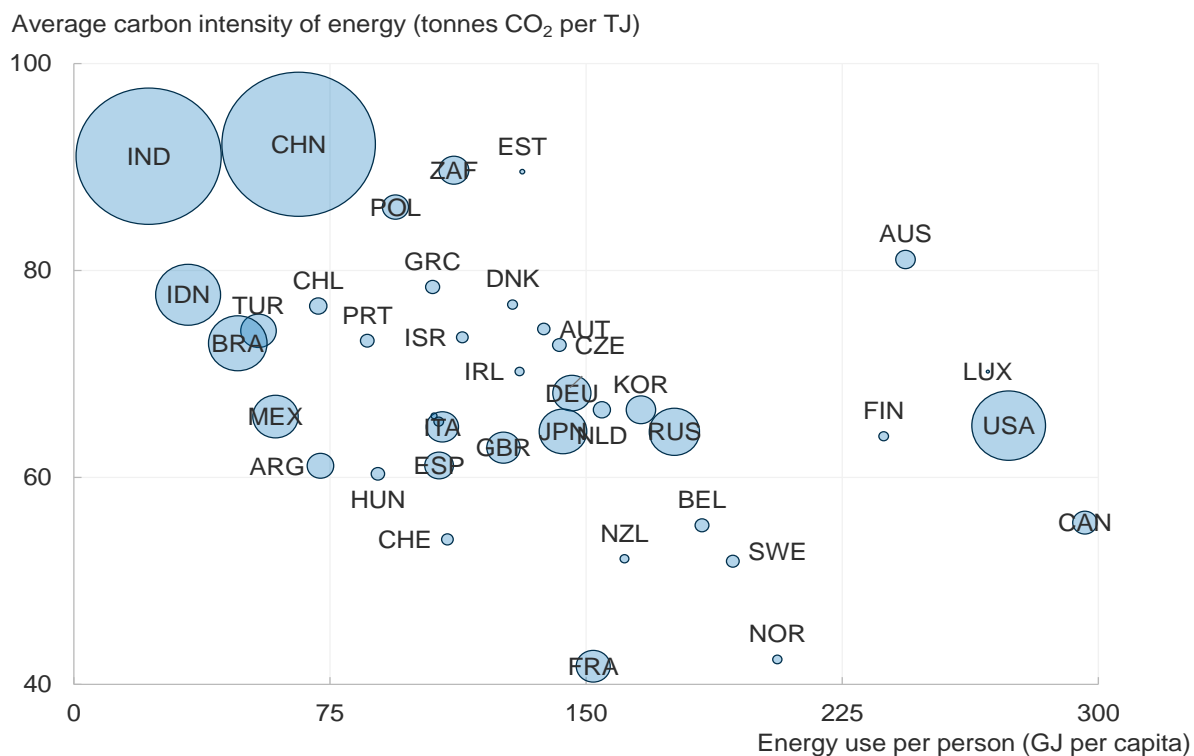
Figure 11: Effective tax rates on gasoline and diesel for road use



Source: Based on OECD (2015b)

Figure 12 points towards some of the challenges ahead. A few very large emerging countries are at present among the more carbon-intensive ones. Environment and climate policy objectives imply that it will be crucial to reduce this intensity as these economies continue to grow.

**Figure 12: Average carbon intensity of per capita energy use, in OECD and G20 economies**



Source: Based on World Bank *World Development Indicators* (<http://data.worldbank.org>).

To conclude this subsection, it appears that energy taxation is shaped by a mix of environment and revenue-raising considerations. Environmental concerns are affecting the policy mix more now than in the past. However, revenue-driven policies still dominate the energy-taxation landscape. This results in considerable misalignment of rates with environmental objectives, and also in considerable tax revenue foregone, especially when considering the very low rates outside of road transport and on highly-polluting fuels. Competitiveness and equity considerations clearly also shape energy taxes, and are important drivers of foregone revenue. The argument made above, that if compensation is needed it preferably should not be delivered in ways that blunts the environmental effectiveness of policies used, is particularly relevant in this sector. In times when revenue considerations provided the main impetus for energy taxation, providing compensation through lower rates may have been an acceptable (if still not ideal) delivery mechanism, but with the increasing weight of environmental concerns, it becomes much less defensible.

Carbon taxes and emissions trading more often come with earmarking or at least strong statements of policy intent regarding revenue use when compared to excise taxes. It is difficult to assess what the social value of these types of revenue uses is, but in general it appears that revenue is deployed in ways that maintains the strength of using price signals to reduce emissions.

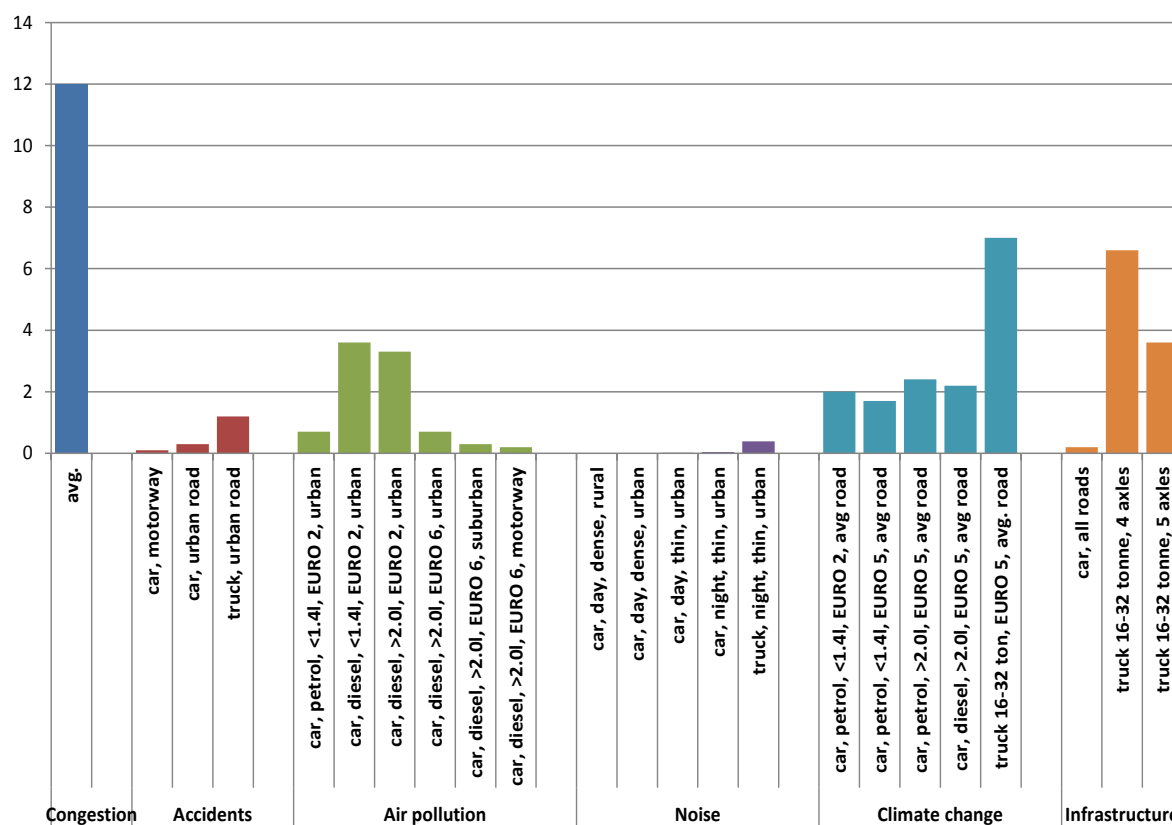
### 4.3 Taxing road transport

The main external costs in road transport relate to congestion, accidents, air pollution, noise, climate change, and wear and tear on infrastructure. Many of these costs, apart from climate change, are more closely related to driving than to fuel use, suggesting that a transport tax system that is well-aligned with external costs should focus on driving-based taxes. Only for emissions of CO<sub>2</sub> are fuel taxes the ideal instrument; for the other costs they are an indirect tax of limited to very limited effectiveness. The question of whether current fuel taxes are roughly at the right levels given the absence of sophisticated driving-

related taxes in most countries, is difficult to answer. Taking the view that fuel taxes should reflect average congestion costs, Parry and Small (2005) find that fuel taxes in the USA are currently too low, and too high in the UK. Newbery (2005) suggest the prices are about right in some European contexts and too low in others.

Figure 13 shows that congestion costs, here averaged over a broad range of driving conditions, are typically the largest external cost. Air pollution, climate change and infrastructure costs can be high too, but much depends on vehicle types (gasoline or diesel, car or truck, new or old, etc.) and driving conditions (urban or rural, congested or not, highway or local road, etc.). This large context-dependence of external costs indicates that even very sophisticated transport taxation systems will not be able to capture all the variation in costs, suggesting the need for complementary instruments (e.g. emission regulation for air pollution). Also, a shift towards driving-related taxation would increase the cost of tax collection. The cost of electronic toll collection is found to be around 20% of revenues at present. Forward-looking estimates suggest this can be reduced to 10% or 5%, which is still more than the cost of collecting fuel taxes, suggesting that shifting towards driving-based charges should come with markedly better management of external costs for it to compensate the higher collection costs (unless, of course, driving-based charges come on top of fuel taxes instead of partly replacing them).

**Figure 13: Main external cost estimates of car use for EU countries - average congestion cost (Euro-cent per vehicle-km, 2010)**



Source: based on Ricardo – AEA (2014)



In European countries, there is a gradual move towards the adoption of distance-based charges for heavy goods vehicles, and perhaps in the future also for passenger cars. Table 1 shows that the introduction of distance-based charges has resulted in higher total taxes or charges for a “representative trip” (in terms of distance, truck type, load, etc.) and not so much in a shift from diesel taxes towards distance-based charges. For example, in Switzerland, distance-based charges were introduced in 2001. Comparing the total tax and charges of a representative trip in Switzerland shows an increase by a factor of 3.62. About 78% of the charge was distance-based in 2008. This pattern is seen, albeit to a smaller extent, in Austria, Germany and the Czech Republic. France, the United Kingdom and Belgium did not introduce new distance-based charges and there was no increase in the total charge between 1998 and 2008. This suggests that revenue interest appears to drive the policy change (with distance-based charges bringing in more revenue than the fuel tax base, which is more prone to tax competition). The challenge for environmental policy-making hence could be to take the opportunity afforded by the introduction of distance-based charges to make sure that the charging system also becomes environmentally more performant.

**Table 1: Level, change and composition of charges and taxes for a “representative trip by truck” in selected EU countries**

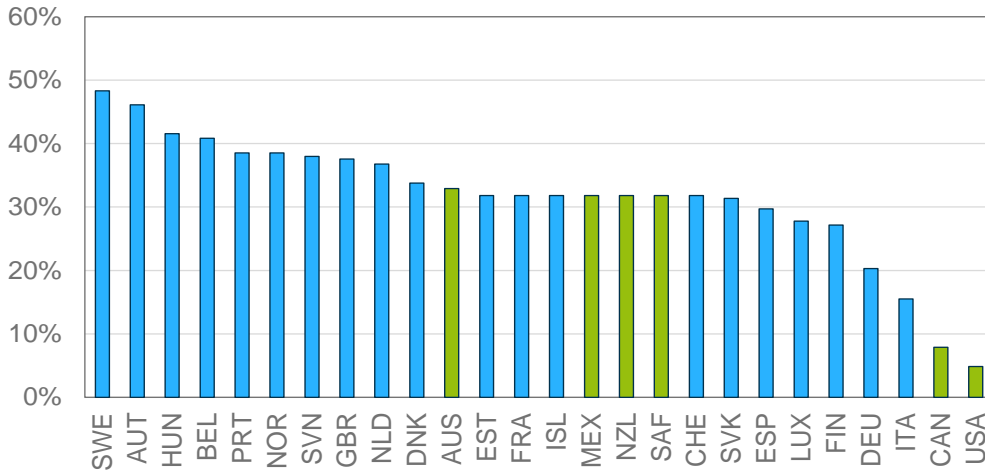
	Electronic charge introduced in	2008 charge/1998 charge	% of total charge that is distance-based
Switzerland	2001	3.62	78
Austria	2004	1.29	58
Germany	2005	1.49	59
Czech Republic	2007	2.05	28
France		0.97	46
UK		0.95	0
Belgium		0.85	0

Source: Own calculations based on ITF data (<http://www.itf-oecd.org/road-haulage-charges-and-taxes>)

Environmental fiscal reform in the road transport sector is not simply a matter of better aligning taxes with marginal external costs. Other important elements of policy reform include the removal of environmentally unproductive subsidies. Examples include, in many countries, the deductibility of commuting costs from income taxes, and the favourable tax treatment of the use of company cars for personal transport. Figure 14 shows the share of company cars in total new car registrations between 2009 and 2011, which is high in many European countries. Company car systems are often treated favourably under personal income taxation in the sense that taxes are lower than they would have been if the value of the company car and its use would have been made available as regular income. Figure 15 illustrates the fiscal downside of several of these company car systems, showing that some countries tax company car value almost as highly as regular income, while others capture less than 20% of potential income. Some systems even make it costless for users to drive their cars more. In addition to revenue implications, such systems result in increased driving and the adverse environmental consequences<sup>14</sup> that come with it.

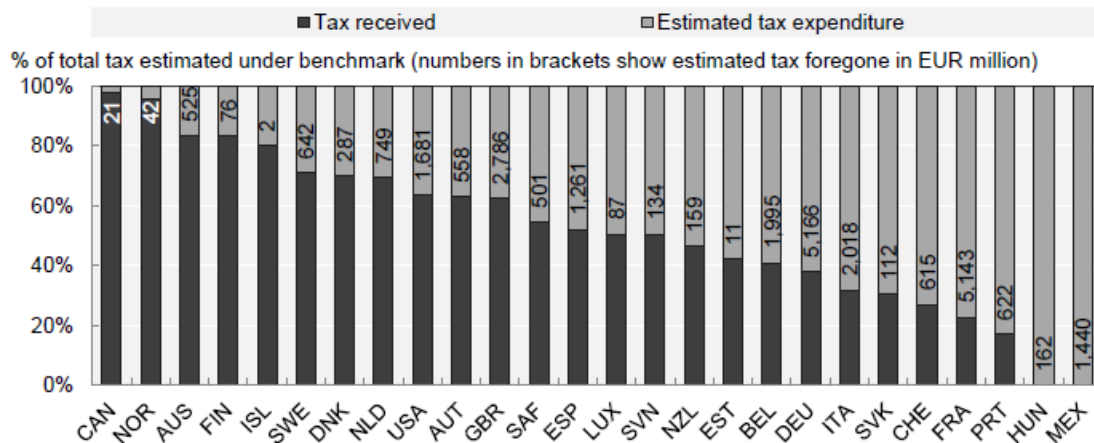
<sup>14</sup> Buidling on Harding (2014), Roy (2014) finds that the environmental and social cost related to the preferential tax treatment of company cars results are larger than would be expected by the size of the company car market.

Figure 14: Share of company cars in total new car registrations, 2009-2011



Source: Harding (2014a)

Figure 15: Estimated tax expenditure as a proportion of total tax under benchmark estimates, 2012



Source: Harding (2014b)

Concluding this subsection, it can be observed that taxes and charges on road transport are generally not very well aligned with the different external costs of road transport. Revenue-raising motives strongly shape pricing instruments. Improved reflection of environmental considerations in policy design, however, does not necessarily run counter these motives. Some tax policies in the road transport sector, generally favouring specific economic sectors or industries, such as the reduced tax rates levied on diesel cars and the treatment of commuting expenses and company cars under income taxes, result in substantial amounts of revenues foregone and have adverse environmental consequences at the same time.

#### 4.4 Taxing and charging for waste

Waste taxes and waste charges can aim to change waste-disposal practices, for example by moving waste “up the chain” from landfilling to incineration, or towards recycling or composting, by setting landfill and incineration taxes respectively. Attempts to connect production and marketing decisions to end-of-life disposal costs include advance disposal fees and extended producer responsibility (EPR) arrangements. Household waste charges (e.g. PAYT) and deposit-refund schemes attempt to change household behaviour, to reduce waste generation and encourage recycling. In general, it appears that

increasing use is made of charging to cover waste management costs, but this practice does not always combine with strong incentives to reduce waste, or treat it differently (OECD, 2016b).

Landfill taxes, typically levied on the weight or volume of waste delivered to landfill sites or on authorised landfill capacity, are relatively popular, in particular in EU countries (Smith, 2008). Cross-sectional analysis of EU countries suggests that higher taxes on landfill are negatively correlated with the proportion of municipal solid waste that is landfilled (Watkins et al., 2012). Deeper analysis suggests that this correlation may not be causal, though, at least not for the countries with the highest tax rates. This is since many EU members with high tax rates also implemented bans on landfilling, and oftentimes rates were increased only after bans were legislated, to discourage companies from lobbying for exemptions from bans (Watkins et al., 2012). While this does not mean that taxes cannot be effective in reducing landfill, it suggests that they succeed in pushing waste up the hierarchy up to a certain point (also in countries without bans), but that further reductions in landfilling activity are mostly due to regulation. Of course, levying taxes at prohibitively high rates might achieve much the same effect as bans, but the non-existence of such high rates suggests that most taxes on landfill are also meant to raise revenues.

Taxes on waste incineration are much less common than taxes on landfill, and, in the EU, just 6 countries operated such instruments as of 2012. Views on the necessity of levying taxes on waste incineration vary. Some sources emphasise that taxing only one route of waste disposal (i.e. just landfill or incineration) encourages excessive resort to the other route (Smith, 2008). Others argue that, since incineration tends to be a relatively expensive treatment option, the amount of waste incinerated is driven by the extent to which high taxes on landfill make incineration cost competitive with landfilling, or whether bans on landfill are in place (Watkins et al., 2012). Across European countries, the observed correlation between total charges on incineration and the proportion of waste incinerated suggested is relatively weak, but there are only few observations. In addition, all EU member states that operate incineration taxes also have taxes on landfill in place, and taxes on landfill are always higher than incineration taxes, which can be interpreted as an attempt to tilt the balance in favour of incineration (ibid.).

Evaluations of pay-as-you-through (PAYT) systems are conducted on a case-by-case basis, since implementation is far from universal and often takes place at the municipal level, with widely varying design features. While evidence suggests that, despite their higher administrative and infrastructure cost, weight-based systems, followed by volume- and frequency-based systems, are more cost-effective in preventing waste, most PAYT systems implemented in Europe levy flat charges. To be effective in discouraging waste generation or increased sorting, PAYT charges need to be relatively high and need to be combined with strong supporting policies. Absent those, the likely outcome is small effects and more illegal dumping. As a result, though they can be effective at pushing waste up the hierarchy, the effectiveness of PAYT systems strongly depends on enforcement and social norms (see e.g. Carattini et al, 2016, for an evaluation of a PAYT system in Switzerland).<sup>15</sup>

Deposit-refund systems impose an upfront fee on consumption and subsidise green inputs and mitigation activities. They are most commonly used for beverage containers, lead-acid batteries, motor oil, tyres, various hazardous materials, and electronics. In theory, as the refund avoids littering, deposit-refund systems are more effective at encouraging recycling than PAYT systems, but empirical evidence on their effectiveness remains scarce (Walls, 2011).

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<sup>15</sup> Similarly, extended producer responsibility (EPR) schemes require strong public and private institutions (see OECD 2016b for a detailed overview of EPR practice).

#### 4.5 *Taxing or charging for water abstraction*

Pricing water supply is one way to allocate water among competing uses and to limit wasteful use and consumption.<sup>16</sup> Pricing also generates revenues, which can be used for various purposes including maintaining, renewing and extending water infrastructure and sanitation services. Some distinctive features associated with water appear to have shaped water tariffs and abstraction charges in OECD countries and beyond.

Water prices significantly influence consumption, though evidence of the effects of water prices is very scarce for population groups other than households. Figure 16 depicts a correlation between higher prices and lower water consumption. Using household survey data for 10 OECD countries, Grafton et al. (2008) show that the average volumetric water price is a significant predictor of differences in residential water consumption.

Since water is an absolute necessity, it is often argued that it should not be priced, or at least not at cost-reflective levels (Hanemann, 2006; OECD 2015c). Concerns about the distributional effects of water prices and of the affordability of water across population groups drive such arguments. Combined with the cost structure of water provision (high upfront investment costs, significant economies of scale and high transport costs) this helps explain the significant public presence in water supply and sanitation and ubiquitous provision of water at prices below or far below production cost. However, certainly in countries with well-developed social security systems, equity goals can be better achieved through instruments that are independent of the amount of water consumed.

Some economic literature argues that water should be priced at the long-run marginal cost of supply (LRMC), reflecting the full economic cost of water supply, the sum of the transmission, treatment and distribution cost, some portion of the capital cost of current reservoirs and treatment systems, and the opportunity cost in both use and non-use value of water for other potential purposes (Olstead and Stavins, 2009). Full reflection of water scarcity in prices means that charges vary over time and space, i.e. over the course of the year, and by the source from which water is abstracted. LRMC pricing, however is not always practical (e.g. it does not ensure that utilities can accumulate sufficient funds for investment, it requires full metering of consumption, and – importantly – it assumes that capacity is fully divisible, an assumption that does not fit the frequent lumpiness, durability and irreversibility observed in reality; Andersson and Bohman, 1985), and only Australia reports LRMC pricing as a principle for setting water prices (Italy and Mexico mention MC pricing as a guide to set tariffs for industry). Most other countries resort to average incremental cost pricing, with the fixed costs for infrastructure recovered as a lump sum (ibid.).

Where they exist, abstraction charges tend to be a mix of cost-recovery elements through fixed charges (e.g. one-time connection fees, a minimum or recurrent fixed charge that can be uniform or linked to some customer characteristics), combined with variable elements (e.g. continuous or block-rate increasing or decreasing volumetric charges, or other variables such as the area of industrial estates; OECD, 2010). In many countries, water abstraction charges also vary by sector, and variation can be large. The residential sector is most often priced and tends to pay the highest charges, water use in industry is less often priced and sometimes at lower rates, and water use by agriculture – frequently the dominant consumptive water user in OECD countries – is often fully exempt from abstraction charges (OECD, 2010a; OECD, 2015). Abstraction charges rarely reflect scarcity, except for example in the two Belgian

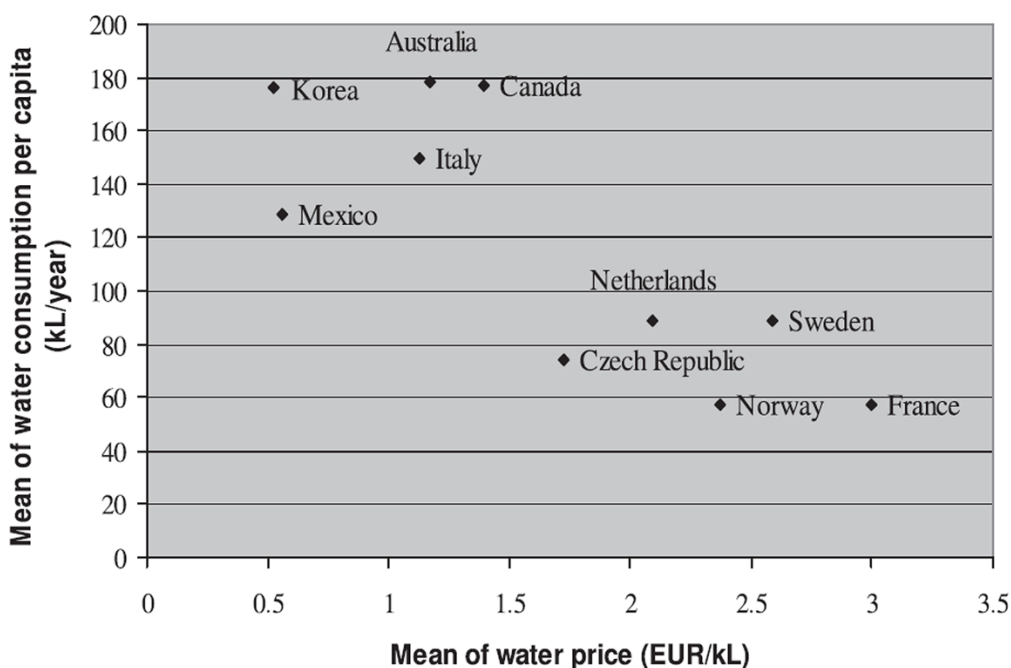
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<sup>16</sup> Please refer to OECD (2010a) and OECD (2015c), for a full treatment of water pricing and allocation systems, including the Annexes for detailed descriptions of policies for all OECD countries. The OECD Environment Outlook to 2050 (OECD, 2012) highlights that water is over-used – i.e. current abstraction levels exceed the sustainable level – or over-allocated – i.e. existing water entitlements to abstract water exceed the sustainable level – in many places around the world. This is compounded by the effects of climate change, increasing water risks, disrupting freshwater ecosystems and significant shifts in the timing, location, amount and form of precipitation future.

regions and the subnational charges in Spain and Israel (OECD, 2010; OECD; 2015c). Recent droughts in countries which have typically not suffered from water scarcity (e.g. France, the Netherlands, Brazil or Sweden) forced governments to resort to relatively expensive measures, such as usage restrictions or water transport over land. Interest in reflecting scarcity in price and charging systems for water could rise in the future.<sup>17</sup>

In most countries, abstraction charges provide funding for water resources management or for watershed production activities, but their low levels mean that they often do not cover costs (OECD, 2010a). In most cases, charges are collected and revenues are retained locally. In Denmark and Mexico, abstraction charge revenues accrue to the general budget.

**Figure 16: Residential water consumption per capita and the mean water price in 10 OECD countries**



Source: Grafton et al. (2011), based on data from a 2008 OECD survey on household water consumption

#### **4.6 Taxing and charging for water pollution**

Taxing water pollution from non-point sources can be complicated in practice, in particular when there are multiple small polluters and when different pollutants are mixed. Estimating firm-level emissions and their damages may be prohibitively costly, due to the presence of multiple small producers, complex chemical processes associated with the different pollutants, and stochastic environmental factors, such as weather (Olmstead, 2010).

Taxes on water pollution are not very common, and countries tend to tax the inputs to pollution instead. Where they are used, e.g. Australia, Canada, France, the Netherlands, Sweden, China, Malaysia and Colombia, design approaches vary widely. Case study evaluations suggest that most such policies do not tax emissions at levels that would approach the level of the marginal damage, but levels are sometimes

<sup>17</sup> See Olmstead and Stavins (2009) for a detailed comparison of the cost associated with price and non-price approaches to water conservation. Some case studies of how countries have responded to local droughts are included in OECD (2015c). Grafton and Ward (2008) and Ward et al. (2011) also discuss the welfare effects of prices versus rationing water use in different circumstances.

high enough to reduce pollution (Stavins, 2003).

Advances in nutrient pollution modelling might provide an opportunity to tax diffuse pollution outputs directly, rather than taxing inputs as proxies. For example, OVERSEER is a model for farm-scale nutrient budgeting and loss estimation in New Zealand, which also identifies risks of environmental impacts through nutrient run-off and leaching. Using such models, pollution charges could be designed in a way that is more aligned with the amount of pollution generated (OECD, 2017).

#### **4.7 Taxing chemicals**

Where implemented, taxes on chemicals can be divided into different groups: taxes on pesticides, fertilizer and other hazardous chemicals, but – due to the limited empirical relevance of taxes on other hazardous chemicals, only the first two are discussed here.<sup>18</sup> Specific taxes on chemicals have been implemented by a limited number of countries, usually in combination with command-and-control measures, as imposed by the hazardous nature of some substances. Relying on regulatory instruments only foregoes the dynamic incentives that market-based instruments can provide.

While farmer responsiveness to taxes on chemicals appears to have been limited so far, new technologies could increase the elasticities of chemical use, as they may facilitate a more targeted application of products. For example, wireless nano-sensors already being used to control and automate the application of fertilisers and pesticides in agriculture (OECD, 2010). Taxing chemical inputs is a workaround to the challenges of administering and monitoring taxes on pollution from non-point sources.

##### *Taxing fertilisers*

Fertilisers are a major contributor to nutrient losses from agricultural soils into ground and surface water bodies, a cause of eutrophication – excessive richness of nutrients in water (Eurostat, 2012). The past decades have seen the issuance of a range of regulations to foster more sustainable fertiliser use, for example the Nitrates (1991) and Water Framework Directive in the European Union. A range of Northern European countries had implemented taxes on fertiliser and its main ingredients – nitrogen and phosphate, but of these, just the Danish tax remains in place as of 2016 (Table 1). In some countries, abolishing taxes on fertiliser was related to the obligation to adhere to EU standards, which taxes by themselves cannot ensure.

With the exception of the Swedish and the Dutch taxes, a strong initial motivation to introduce a tax on fertiliser was to fund other policy measures, predominantly to the benefit of agriculture. Environmental considerations also affected tax introduction and design. Evaluations of the behavioural effects of fertiliser taxes mostly resort to correlations between tax levels and product use. While some studies associate taxes with use reductions, it remains difficult to discern their effects from those of competing regulations, standards, and awareness raising campaigns. Environmental impacts are even more difficult to pin down, since the environmental damage of fertiliser varies strongly with the receiving environment at local level. Nonetheless, some improvements in groundwater quality are associated with the taxes on fertiliser implemented in The Netherlands and Sweden (e.g. Westhoek et al., 2004). In general, it is argued that tax levels have been too low to foster significant changes in use or environmental impacts.

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<sup>18</sup> Few tax instruments on hazardous chemicals exist. Australia, Denmark and the US levy taxes on ozone-depleting chemicals, but there is a dearth of evidence on their effectiveness in reducing the use of these substances. This is mainly due to instruments which were implemented in parallel (e.g. a tradable permit system, in the case of the US), and the Montreal Protocol, effective in 1989, which is credited for being very effective at reducing and reversing the decrease of ozone-depleting substances in the atmosphere. Chlorinated solvents, used in degreasing and dry-cleaning, are taxed in Denmark and Norway. Both countries experienced considerable decline in the use of these chemicals after the taxes were introduced, between 25 and 76%, depending on the product, in Denmark and 82-90% in Norway.

The Dutch tax system (Mineral Accounting System - MINAS) is noteworthy in how it combined regulation with taxes to target *excess* fertiliser use. Until 2005<sup>19</sup>, MINAS taxed nitrogen and phosphate use in excess of a certain levy-free quantity per hectare, which was based on EU environmental standards and levied across almost all farm types. The tax targets ‘wasteful’ use, for which the elasticity of the tax base likely is higher than for use in general, and which is also less likely to affect farmers’ competitiveness. Targeting excess use also appears sensible since, while excessive fertiliser use causes pollution, positive application rates are required to replace nitrogen and phosphorus lost through intensive cropping (Eurostat, 2015).

**Table 2: Overview of taxes on fertiliser implemented to date**

	Year of introduction	Abolished in	Tax design	Revenue use	Environmental objective
Austria	1986	1996, upon joining the EU	ATS 3.5 per kg nitrogen ATS 2.0 (~EUR 0.15) per kg phosphate	Support grain production sector through export subsidies	Soil conservation
Denmark	1998	-	DKK 5 per kg nitrogen, with broad exemptions for agriculture	Reductions in land use tax	Reduce nitrate pollution
Finland	1976	1995	Revised multiple times, level lower than in Denmark and Austria	Finance export subsidies	From 1990: reduce phosphorous content of fertiliser
The Netherlands	1998	2005	Tax per kg nitrogen and phosphate in excess of a regulated threshold	Feeds into general budget	-
Norway	1988	2000	<i>Ad valorem</i> for nitrogen-based fertilisers, gradually increased from 1% to 20% in 1991	Finance environmentally-friendly cultivating practices and information measures	-
Sweden	1984	2010	~20% of the fertiliser price	Reduce negative impacts of chemical use in agriculture, finance R&D measure for agriculture	Reduce chemicals leakage into soil

Source: Based on Söderholm (2009), updated using the OECD Database on Policy Instruments for the environment (<http://www2.oecd.org/econstat/queries/Default.aspx>)

### *Taxing pesticides*

Pesticides are a relatively heterogeneous group of products, the most common broad groupings span fungicides, insecticides, herbicides, plant growth regulators and molluscicides (which target snails). Herbicides and fungicides account for the major share of pesticide sales in the EU-28 (Eurostat, 2016). Pesticide use affects both the receiving environment (e.g. groundwater pollution) and animal

<sup>19</sup> This system was abandoned after the European Court ruled that taxation based on surpluses over a specified level was not compatible with the EU Nitrates Directive, and that taxes should instead be based on nutrient inputs (Hogg et al., 2014).

and human health (e.g. poisoning of agricultural workers, unwanted residues in food and water). In particular, exposure to pesticides through residues may leave people more susceptible to diseases from viruses and bacteria, and there is an on-going debate on whether some pesticides might be carcinogenic.

Regulation (e.g. prohibition or periodic licensing of products) has a strong role to play in reducing pesticide use, but taxes can play an important steering role to foster more efficient product use. Well-designed price-based instruments create on-going incentives to foster substitution with less harmful products (also below maximum allowable quantities) or incite innovation (e.g. move towards replacing the products listed on the European Union list of active substances “candidates for substitution”).

Taxes on pesticides are currently in place in five countries (Table 2). Revenues are often earmarked at the time of introduction, but Sweden, Denmark and Norway moved away from it. Environmental considerations appear to play a larger role for the introduction of these taxes than is the case for fertilisers. Some taxes on pesticides are implemented as *ad valorem* taxes, e.g. the new tax in Mexico.

Taxing on a per-volume basis of active substance does not differentiate for pesticide load, and ignores that some newer products are effective at lower quantities though they can be more toxic (Söderholm, 2009). Most countries levy specific rates per kg active substance (e.g. Sweden, France). Norway and Denmark have very differentiated systems, which reflect the products’ toxicity for humans and the environment. Of all taxes in place, the Danish tax design is the most sophisticated, as it augments a base rate per active substance by its toxicity for human health and the environment, but also its degradability in soils, bioaccumulation and leaching potential (“environmental fate”).

The effectiveness of taxes on pesticides in reducing the use of harmful products is difficult to establish, due to a the relatively dynamic market and the large variety of products, observed hoarding behaviour before tax increases, large seasonal and geographic variations in treatment intensity and frequency, and the effects of competing regulations. Furthermore, due to the existence of newer, low-dose products, reductions in sales or use quantity are not a sufficient indicator of effectiveness. In fact, after a continuous decline in pesticide sales in the last two decades, pesticide load and treatment frequency appear to have increased again in more recent years (Eurostat, 2015, Böcker and Finger, 2016).

**Table 3: Overview of the pesticides taxes in place, as in April 2017**

	Introduced in	Tax design	Point of imposition	Revenue use	Environmental objective
Sweden	1984	Flat, at SEK 34/kg AS	Producer/importer	General budget	Reduction of residues in surface water or food, reduce environmental risk associated with pesticide use
Norway	1988	25 NOK/ha*(human health and environmental risk indicator) for seven categories of toxicity	Producer/importer	General budget	
Denmark	1996	DKK 50/kg*active substance/litre + DKK 107*toxicity	Producer/importer	General budget	
France	2000	EUR 0.9/kg, EUR 2/kg, EUR 5.1/kg, depending on toxicity.	Retailer	Compensation measures for farmers	
Mexico	2014	Different ad valorem rates for 5 bands of toxicity, 6-9%	Producer/importer	General budget	

Source: Based on Böcker and Finger (2016)



## 5. Concluding remarks

Environmentally related taxes are used extensively in some sectors of the economy, notably the transport sector, to some extent on energy use, and much less in other sectors. Good practice would require broad coverage, strong and stable prices (rising over time for greenhouse-gas emissions) and socially productive use of the revenues. Current practice is of limited coverage, low and sometimes volatile rates in most sectors except transport, and strong rate differentiation across and also within sectors of the economy. Rate differentiation across sectors can make good sense if the full set of relevant externalities is considered and if revenue-raising objectives are taken into account. However, rate differentiation within sectors as a result of compensation given to some taxpayers is much harder to defend. Similar concerns apply to awarding free permits in emission trading systems.

Environmental and fiscal objectives shape policy design to different extents across policy areas. In the case of energy taxes, revenue-raising objectives often dominate tax design, in particular in the case of taxes on transport fuels. Environmental considerations play an increasing role in the energy and carbon tax realm, even if explicit and implicit revenue use considerations often shape policy design very strongly. Distance-based charges and taxes on transport are shaped relatively strongly by revenue-raising motives, though environmental considerations do play a role, and could play an increasing role in policy design.

Pricing instruments for waste are very diverse, and are primarily introduced to transform waste management practices and for cost-recovery. Water prices and charges are usually low, too low to recover costs, and existing pricing instruments, do not reflect water scarcity very strongly. Taxing water pollution directly is very difficult, and some countries resort to taxing chemical inputs instead. These taxes interact very strongly with regulations, and some are based directly on it.

Concerns about the protection of domestic industry shape tax design in very similar ways across policy areas, and the large polluters often are exempted from taxes. Distributional considerations appear to shape policy design less strongly than industry protection.

Environmental fiscal reform holds considerable promise. It potentially is the cornerstone of cost-effective environmental policy, but with few exceptions it does not deliver on that promise yet. Equity and competitiveness considerations should not be overstated, and where they are relevant they can be addressed in ways that do not compromise the environmental effectiveness of market-based instruments. Environmental fiscal policy also holds considerable potential in the context of fiscal policy. Revenue interests are real and can be combined with effective environmental policy. In fact, increased awareness of the revenue potential can help ensure that poor revenue use choices are avoided, perhaps resulting in more “explicit revenue”, i.e. more revenue raised straight from the taxes or permit auctions, with less preferential treatment and thus less foregone revenue. Such a shift would further reinforce further the environmental effectiveness of market-based instruments.

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