# The French Tax on Air Pollution: Some Preliminary Results on its Effectiveness

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**Summary**

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**Keywords**: Environmental tax, emissions regulation, earmarking, air quality

**JEL**:H00, H23, Q25, Q28



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**Abstract**

Empirical evidence evaluating the efficiency of economic instruments is still rare, despite significant theoretical advances over the last decades. The objective of this paper is to evaluate one form of environmental taxation, the French tax on air pollution from 1990-99. While starting out in 1985 as a tax levied only on emissions of sulphur dioxide ($SO_2$), the tax base was subsequently extended to encompass also emissions of nitrogen oxides ($NO_x$), hydrochloric acid (HCl), and volatile organic compounds (VOC). The revenues of the French tax on air pollution were earmarked for abatement subsidies and the financing of air quality surveillance systems. Using a plant-level database, we find a negative, significant effect of the tax on emissions of $SO_2$, $NO_x$, and HCl. The abatement elasticity with regard to the tax is quite small, however.

## 1 Introduction

Empirical evidence on the actual effects of economic instruments is rare, despite significant theoretical advances over the last decades on the efficiency of economic instruments such as taxes or tradeable permits. One exception is the recent policy lesson drawn from the market in tradeable sulphur permits that was introduced by the United States Clean Air Act Amendments in 1990 (Joskow and Schmalensee, 1998; Joskow, Schmalensee and Bailey, 1998 ; Stavins, 1998). It has also been possible to measure the efficiency of the permit market created for the lead phasedown in the United States between 1982 and 1987 (Kerr and Maré, 1996). Most of these evaluations concern the United States, though, whereas Europe, for instance, has relied to a larger extent on environmental taxes. The question of assessing the impact of environmental taxes is complicated. First, environmental taxes in Europe have almost always been set at a level lower than that justified by environmental damages, and have rather been used for revenue-raising purposes. Is it fair to evaluate the environmental impact and economic efficiency of an instrument whose primary purpose is to raise revenue? Second, environmental taxes do not operate in isolation. In Europe, they usually complement traditional regulations by technology standards and are in many cases used to speed up the introduction of new technology. How to entangle the effect of environmental taxes from other forms of regulation simultaneously in use represents a major difficulty. Third, the necessary data (preferably plant-level) is hard to find.

The purpose of this paper is to evaluate one form of environmental taxation, the revenue-recycled air pollution charge that was in use in France from 1985-1999. This is one of few actual ex post analyses of specific environmental taxes, with the exception being the detailed evaluation of the Swedish emissions charge on nitrogen oxides by Höglund (2000) and Sterner and Höglund (2000).[1] The paper presents an econometric analysis based on plant-level data of emission levels for four major air pollutants: sulphur dioxide ($SO_2$), nitrogen oxides ($NO_x$), volatile organic compounds (VOC) and hydrochloric acid (HCl). The French air pollution tax has been interpreted as an example of a revenue-raising instrument, of which the revenues were allocated to abatement subsidies and the financing of air quality surveillance. However, despite the relatively low level of the tax (as compared to charges implemented in Scandinavia, for example) the abatement obtained from the recycled revenues may lead to a positive impact in terms of reduced emission levels. In this paper we situate the French air pollution tax in its regulatory context, review the related literature, and present some preliminary results on the impact of the tax.

## 2 Regulatory Framework

Regulation of air pollution is significantly affected by the international Convention on Long Range Transboundary Air Pollution, which was signed in 1979 following intense debate on acid deposition caused by transboundary air pollution. Since its entry into force in 1983, the Convention has been extended by several protocols. In 1985, the so called Helsinki Protocol was signed that stipulated a reduction of sulphur emissions (or their transboundary fluxes) by at least 30 per cent. The proportional reduction target was abandoned in 1994 in favour of an approach basing targets on critical loads, where the target load differs according to countries' sensibility to acidification. Over the years, the Convention has been extended to encompass the control of

[1] Brännlund and Kriström (2001) evaluate the impact of the Swedish energy taxation by estimating production functions on a panel of 150 district heating plants, and simulating the resulting changes in emissions of sulphur, nitrogen oxides, particulates and carbon dioxide.

nitrogen emissions (the Sofia Protocol of 1988), volatile organic compounds (1991), heavy metals (1998), and persistent organic pollutants (1998). In 1999, the Gothenburg Protocol to Abate Acidification, Eutrophication and Ground-Level Ozone was signed. The Protocol sets emission ceilings for 2010 for sulphur (measured as $SO_2$), $NO_x$, VOC and ammonia (NH3). In aggregate, the Protocol aims at a reduction for the European Community of 75% of sulphur emissions, 49% of $NO_x$ emissions, 57% of VOC emissions and 15% of ammonia emissions in terms of 1990 emissions and including both mobile and fixed sources.

The international Convention on Long Range Transboundary Air Pollution has been a driving factor for regulatory action on behalf of the European Union, which in turn drives national regulation in France. The most significant directive was issued in 1988 to limit sulphur and nitrogen emissions from large combustion plants with a capacity exceeding 50 MW thermal (Directive 88/609/EEC ). The Directive stipulated emission limits for new plants and national ceilings for total emissions. In the year 2000, agreement was reached on an amendment of the Large Combustion Plant Directive including stricter emission limits for 1 January 2008. Another relevant piece of European legislation is the Integrated Pollution Prevention and Control (IPPC) Directive (96/61/EC) that sets out to reduce pollution from fixed sources using a licensing procedure for each plant based on Best Available Technique.

# 3 Literature Review

## 3.1 Applied regulatory analysis

The existing literature can be classified into three major categories: analyses of national strategic behaviour using game theory, assessments of the macro-economic impact of $SO_2$ and $NO_x$ abatement, and surveys and analyses of existing regulations in different countries.

Murdoch, Sandler and Sargent (1997) focus on the transboundary public good aspect of sulphur and nitrogen oxides emissions. They underline the physical differences of the two pollutants in explaining why the response to regulation has been so different for $NO_x$ versus $SO_2$ emissions. Modeling countries' abatement as a non-cooperative Nash response, they derive voluntary levels of abatement in excess of those stipulated in international conventions on long range transboundary air pollution. In an empirical test, the representation of voluntary emission reductions holds for $SO_2$. For $NO_x$, the authors note that the transboundary effect is larger and that countries thus have a larger strategic incentive at the supranational level. Also, the Sofia Protocol that stipulated reductions in both $SO_2$ and in $NO_x$ emissions did not come into effect until 1991, that is, at the end of the study period, and it did therefore not have a deterring effect on $NO_x$ emissions. Furthermore, it is more difficult to implement $NO_x$ abatement policies since the main source of $NO_x$ emissions is traffic. Sulphur emissions, on the other hand, are more easily controlled since the larger part emanates from publicly owned power plants. For the $NO_x$ problem, the authors conclude that the collective action problem first has to be solved at a national level before being confronted on the international plan.

Alfsen, Birkelund and Aaserud (1995) combine a sectoral European energy model that covers nine European countries with the RAINS emission transport- and deposition model, in order to simulate the impact on emissions of $CO_2$, $SO_2$ and $NO_x$ of the energy/carbon tax proposed by the European Commission in 1992. The abatement costs used in the model of transboundary pollution are from the RAINS model, and represent technical cost measures. They find that the impact of the tax and the deregulation of the power sector are larger on $SO_2$ emissions than on $NO_x$, since 70% of the $NO_x$ emissions in the model area in the year 2000 come from the transport sector, which is unaffected by energy sector deregulation and the European Commission energy tax proposal. When appraising the costs of reaching the targets stipulated in the Helsinki and Sofia Protocols, the authors find that $NO_x$ abatement costs are less elastic than $SO_2$ abatement costs. In particular, Norway and France have high marginal abatement costs for $NO_x$, which is explained by an inflexible energy structure, and those countries would face high total control costs to implement the targets in the Sofia Protocol. The authors underline that the use of economic instruments to control one pollutant has to take into account its effects on other pollutants.

Van Rompuy (1997) simulates the effects of Belgian regulation of $SO_2$ and $NO_x$ emissions in a partial equilibrium model. The regulation imposes three constraints on central electricity producers in order to conform with the EC Large Combustion Plant Directive: a 1% sulphur content limit on fuels used in electricity generation, a $SO_2$-$NO_x$ emission reduction scheme for 1991-2003, and technological standards for new power plants. The author finds that the regulatory costs to Belgium exceed the national environmental benefits, since much of the sulphur and $NO_x$ reduction benefits the rest of Europe. An important result is that the side effects on $CO_2$ emissions can be important (in this case a 51% reduction compared to the no regulation scenario), mainly because the regulation induces a shift away from coal power plants to nuclear energy.

Larsen (1997) evaluates the macro economic effects of $NO_x$ abatement in the Norwegian economy using a disaggregated sector model. The abatement measures studied in the industrial sector are the installation of low-$NO_x$ combustion technology in one of the three major petrochemical plants and the installation of catalytic converters at the major oil refinery. The author confirms the presence of secondary economic effects of abatement measures but concludes that the macro economic effects of the increased investment costs are low for the Norwegian economy in terms of reduction in GDP or private consumption.

For $NO_x$ emissions in particular, Cansier and Krumm (1997) provide a useful survey of the existing economic regulation of $NO_x$ emissions in Scandinavia, France, the Netherlands and Japan. The French and Swedish $NO_x$ taxes are compared; the Swedish tax rate is 100 times higher than its French counterpart. The Swedish reimbursement system is aimed at decreasing the amount of nitrogen emissions per energy unit, whereas the French allocation system of subsidies is less transparent. In both countries, the coverage of emission sources is incomplete, since the transport sector is excluded. Furthermore, the thresholds for taxation create some inefficiency in either system.

### 3.2 Theoretical analysis of output-based refunding

The French tax system that we study here resembles partly the Swedish $NO_x$ charge in that tax revenues are rebated back to industry, although on a different basis. The Swedish charge is automatically refunded based on useful energy whereas the French tax is refunded in the form of abatement subsidies for tax-paying companies. One particular interest of our study is to compare the distributional impact of the French tax compared to alternative refunding mechanisms. We therefore briefly review the main results on output-based refunding.

Fischer (2001a) analyses and compares three variants of output-based refunding of environmental policy revenues: tradeable performance standards, output-rebated emission taxes and output-allocated emission permits. The analysis is done for a representative firm with a fixed emissions ratio from output and an emission reduction cost function that is linear in output. Under output-based rebating of emission taxes, total tax revenue is rebated back to the firms according to their share of industry output. A tradeable performance standard fixes the average emission rate and firms are free to sell any reductions below this standard ; if it produces at an above average emission rate, it instead has to purchase emission permits. Assuming perfect competition, output-rebated emission taxes and tradeable performance standards are equivalent. Although the same marginal incentive for emission reductions holds as under an emission tax, the firm will produce a higher level of output compared to the optimal solution. Consequently, for any given emission rate, tradeable performance standards or output-based rebating of emission taxes induce less emission reductions than would the optimal policy of a Pigovian tax. Similarly, output-allocated permits also raises the marginal cost of emission reductions.[2]

Fischer (2001b) extends the basic model to allow for incomplete competition and imperfect participation. When firms' market shares are significant, the output-based subsidy will give incentives for yet further output increases and the consequence could be a shift in industry structure towards more high-emitting firms compared to an emission tax with a fixed output subsidy. If there also is imperfect competition in the output market, the optimal subsidy to correct for this distortion should be inversely related to the elasticity of demand; with heterogenous firms, however, an output-based rebating system cannot achieve optimality. Leakage effects from unregulated sectors cause further distortion, and in such situations the subsidy implied in an output-based rebating system can either be too small or too large, so that in some cases no subsidy is a dominating policy.

Recent policy attention to output-based allocation of environmental tax revenues stems in part from the success of the Swedish refunded $NO_x$ charge on emissions from fixed sources. Under this system, a $NO_x$ tax is levied on large fixed emission sources[3] who are required to install continuous emission meters, and the revenues are refunded to participants directly based on their final production of useful energy. Between 1992 and 1998, the 200 combustion plants that initially were targeted by the regulation reduced their average emission rates by 40%.[4] Sterner and Höglund (2000) analyse output-based refunding of emission taxes with special attention to the Swedish experience. The authors argue that the output effect empirically has been negligeable in the Swedish case, since the cost share of the charge is low, the demand elasticity of the product is low, and there are large technical abatement possibilities. Since the Swedish scheme rebates tax revenues according to plant production of useful energy, the output in the model is effectively

[2] Fischer (2001a) also discusses some additional disadvantages with output-based allocation of environmental tax revenue: defining output and the relevant sector coverage, and unexpected entry and exit effects following manipulation of eligibility for an output subsidy.

[3] Defined initially as units with a production of useful energy above 50 GWh per boiler. The system was extended in 1996 to include all boilers with a useful energy production above 40 GWh, and from 1997, all boilers with at least 25 GWh of useful energy per year.

[4] Measured in kg NOx/MWh useful energy.

energy, an input into most production processes. It would seem that for a profit-maximizing firm that produces part of its own energy input, the incentive effects of the Swedish scheme are somewhat different from those stipulated under a general output-based refunding scheme. Energy could be seen as a secondary output in a multiple output production process and a limitation of existing analyses is the lack of discussion of this discrepancy between the models and the existing regulation. The system is aimed at encouraging reductions in emissions in relation to useful energy output, and all industrial sectors, apart from the metals industry, have reduced their average emission ratios. Certain sectors are doing better than others under the system; The distributional effect of the charge shows that the energy sector has gained in aggregate and that the paper and pulp sector has been a net payer to the system. That does not account for earlier abatement investments, however, but it suggests that the complementarity between output production and fuel generation may be important in explaining the effect of the Swedish refunded $NO_x$ charge.

The existing literature notes that the effect on output is suboptimal and leads to too many firms in the industry in the long run. However, other dynamic effects of output-based refunding policies are not directly examined, such as its effects on innovation and technology adoption.

### 3.3 Survey of empirical econometric papers

Recent empirical papers dealing with air quality regulation focus mainly on two types of impacts: impact on air quality (Henderson, 1996) and impact on firms through their level of compliance (Nadeau, 1997), choice of location (Henderson, 1996) and economic performance (productivity, employment...) (Greenstone, 2001; Gray and Shadbegian, 1994). The datasets used in these studies have been built following surveys conducted exclusively on American firms.

Henderson (1996) examines effects of ground-level ozone regulation on air quality and economic activity. Contrary to past studies, and using panel data, this author shows that local regulation has non-perverse effects on air quality and on firm location decisions. As explained by Henderson (1996), "the location of polluting activity, high concentration readings, and the designation of nonattainment status are of course all strongly positively correlated, cross-sectionally; so air quality indeed is worse in nonattainment areas,[5] and polluting firms are predominantly found there".

[5] After 1970, the Environmental Protection Agency (EPA) established separate national air quality standards for four pollutants. Counties were thus designed "attainment" or "nonattainment" counties.

Nadeau (1997) looks at the effectiveness of the Environmental Protection Agency (EPA) in reducing the time that manufacturing plants spend in a state of noncompliance. The novelty of this paper lies first in the fact that it focuses on the length of time that plants spend in violation of EPA regulation and second, in the fact that it separates the effects of two types of EPA activity: monitoring and enforcement. Using data on 175 plants in the pulp and paper industry on the 1979-1989 period, Nadeau (1997) shows that the EPA is effective at reducing the time plants spend violating standards.

Greenstone (2001) estimates the effects of environmental regulation on industrial activity, using a broad dataset made of 1.75 million plant observations covering the 1967-87 period. Regulation is measured by the county-level attainment/nonattainment designations for each of the four pollutants targetted by the EPA. Controlling for plant-specific, industry-specific and county-specific effects, Greenstone (2001), using a simple linear static model, estimates the impact of the 1970 Clean Air Act Amendments on growth of employment, investment and shipments of manufacturers. Using a simultaneous-equations model, the author finds that the Clean Air Act Amendments substantially retarded the growth of polluting manufacturers in nonattainment counties.

Gray and Shadbegian (1994) analyse empirically the impact of environmental regulation on productivity using plant-level data for three industries: paper mills, oil refineries and steel mills, for the 1979-1990 period. These authors focus on Total Factor Productivity (TFP),[6] examining both productivity levels and growth rates for each plant, and their relationship to the regulatory measures. They find that plants which spend more on pollution abatement are significantly less productive. One of the difficulties in this analysis is that some unobserved characteristics of the plant (for example the quality of the plant management) might bias the estimates: if a more productive plant is more likely to comply with regulations, this would tend to create a negative correlation between abatement costs and productivity.

Using the same data set, Gray and Shadbegian (2001) study differences in the impact of environmental regulation across different plants within a single industry (the paper and pulp industry here). They find a significant negative relationship between pollution abatement costs and productivity levels, which is almost entirely due to mills which incorporate a pulping process.

A common feature of these papers is that they point out that the impacts of air quality

[6] TFP is measured as the difference between output (measured by the value of shipments) and the weighted average of inputs (labour, material and energy expenditures, and capital stock).

regulation have to be studied at a local level and on an industry-by-industry basis.

## 4 The French Tax on Air Pollution

Whereas previous plant-level econometric analyses concern US data almost exclusively, our analysis is done on panel data on firms subject to the French tax on air pollution.[7] The French tax on air pollution (la taxe parafiscale sur la pollution atmosphérique, or TPPA) belongs to the special tax category of parafiscal taxes that do not need approval by parliament and whose purpose is to benefit the payers of the special tax, by definition an earmarked tax. One of its original purposes was to finance investments in air quality surveillance systems. However, its creation was also inspired by the French water pollution charges which are levied upon industrial water users and then recycled in the form of abatement subsidies. The event that spurred its implementation was the European debate on acid rain. In the 1980's, it was clear that policy action was demanded on a European level. France, on the other hand, could do relatively little to reduce its sulphur emissions per unit of energy use, since it had already reduced emission levels through its investment in nuclear energy. Additional incentives were sought to motivate industry to undertake further abatement action.

Initially introduced in 1985 for $SO_2$ emissions, the government order on the TPPA was extended in 1990 to include also emissions of $NO_x$ and HCl, and in 1995 to include VOC.[8] The tax was imposed from 1990 on any entity that fulfills either of two criteria: a maximum combustion capacity equal to or exceeding 20 MW or annual emissions of more than 150 tonnes of either $SO_2$, $NO_x$, HCl, or VOC. Household waste incineration plants with a capacity exceeding 3 tonnes an hour were also subject to the tax.

In 1990, the tax rate targeting $SO_2$, $NO_x$ and HCl emissions was put at a level of 150 FF/tonne.[9] It was increased in 1995 to 180FF/tonne for $SO_2$, $NO_x$, HCl, and VOC and in 1998 to 250FF/tonne for $NO_x$ and VOC. If the total tax due was less than 1,000 FF for a unit, no tax was levied. In 1997, this made for a total of 1,454 tax payers.

The revenue from the TPPA was earmarked for subsidies to abatement or for preparatory

[7] For other evaluations on European data, see the recent paper by Brännlund and Kriström (2001), in which they estimate production functions on panel data for 150 Swedish district heating plants for the period 1989-1996 and simulate the impact of changes in energy taxation.

[8] The extension in 1995 also included small particulate matter, but the tax rate was set at zero for those emissions. For administrative purposes, the inclusion of particulate matter allowed companies to apply for abatement subsidies also for such emissions. We will not consider this particular pollutant in the empirical analysis.

[9] Approximately USD 23/tonne.

technical studies (corresponding to 75% of the tax revenues), with the rest aimed at investment in air quality surveillance systems. The tax was administered by the French Agency for Environment and Energy Management (ADEME)[10] which received 6% of the tax revenue for its administration costs.

The system was based on self-reporting of emissions from the previous year by 15 April. ADEME reports a high level of enforcement: over 90% of taxes due were actually paid. The TPPA in fact raised two major controversies. First, upon its set-up, industry contested the distribution of the tax revenue to investment in air pollution surveillance systems, arguing that all of the tax revenue should be recycled in the form of abatement subsidies. Second, and more important for estimation purposes, major chemical companies contested that $N_2O$ counted as an air pollutant according to the (old) Law on Air Pollution from 1961, and consequently refused to pay for those emissions, only paying for NO and $NO_2$ emissions.

In the year 2000, the TPPA was replaced by a general pollution tax[11] levied by the customs authorities and no longer administered by ADEME, who nevertheless continues to handle requests for abatement subsidies paid out of the general government budget. Our analysis thus encompasses the period when the TPPA was an integrated earmarked tax system.

# 5 Descriptive analysis

Data have been provided by ADEME. This database includes all plants subject to taxation following air regulation in France. 1,942 plants have been recorded between 1990 and 1999. For each plant, information is given on its geographical location, the activity sector it belongs to, the total amount of emissions for four pollutants ($SO_2$, $NO_x$, HCl and VOC), the total amount of taxes paid to the agency, and the maximum combustion capacity (in MW). All monetary amounts are in FF1990. The index used is the price of sales of industrial products (source: INSEE, the French National Statistical Institute).

In Figure 1, we report total emissions of VOC and HCl (in tonnes) and total emissions per unit of combustion capacity (emissions are in tonnes/MW) of $NO_x$ and $SO_2$ for each year between 1992 and 1998.[12] Since $NO_x$ and $SO_2$ emissions originate mainly from the combustion of fossil fuels, we work with emissions per unit of combustion capacity in order to correct for

[10] Agence de l'Environnement et de la Maîtrise de l'Energie.
[11] La Taxe Générale sur les Activités Polluantes (TGAP).
[12] In 1995 emissions were recorded for a six-month period only and have thus been readjusted.

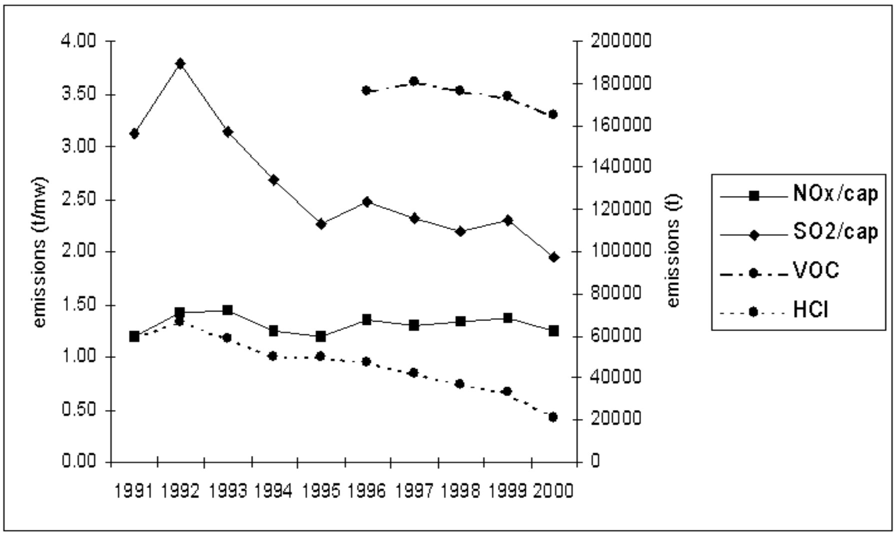


Figure 1: Total emissions of the four pollutants for the period 1990-1999

size effects. Note that emissions of VOC only were taxed, and thus reported, from 1995. The largest emissions (in tonne/MW) are $SO_2$ emissions, followed by $NO_x$. In absolute levels, VOC emissions are greater than HCl emissions. We note a slight increase of all types of emissions in the beginning of the period (in 1992), followed by a net decrease in the case of $SO_2$ emissions, a slight decrease in the case of HCl and VOC whereas emissions of $NO_x$ in terms of tonnes per capacity remained almost constant all along the period.

In Table 1, we report total emissions for the period by sector. Most of the $NO_x$ emissions come from the sector of electricity production (28%) and from the chemical industry (23%). The third big polluter is the sector of other nonmetallic products, accounting for 13% of total $NO_x$ emissions. Main emitters of $SO_2$ are the sectors of coke (30% of total emissions) and electricity production (26%). For HCl emissions, the waste management sector is the main contributor to emissions (62% of total emissions). The two other sectors emitting HCl are electricity (16%) and heating (13%). For VOC, emissions are due in almost equivalent proportions to the sectors of chemistry (20%), plastic and rubber (21%), coke (15%) and the car industry (14%).

As explained in Section 4, the revenue from the taxes is earmarked for subsidies to abatement or for technical studies. We will consider from now on subsidies to abatement only. In Figure 2 we report the number of granted subsidies per year and their total monetary amount in millions

Table 1: Total emissions (in tonne) for the 1990-1999 period by sector

| Sector | $NO_x$ | | $SO_2$ | | HCl | | VOC | |
|---|---|---|---|---|---|---|---|---|
| | Total | % | Total | % | Total | % | Total | % |
| Extraction | 19,695 | 0.5% | 58,136 | 0.8% | 4,532 | 1.0% | 329 | 0.0% |
| Food & agri. prod. | 114,950 | 3.0% | 344,528 | 4.5% | 3,798 | 0.8% | 21,478 | 2.3% |
| Textile | 8,241 | 0.2% | 31,882 | 0.4% | 480 | 0.1% | 13,952 | 1.5% |
| Paper | 92,520 | 2.4% | 235,672 | 3.1% | 3,700 | 0.8% | 55,041 | 5.8% |
| Printing | 636 | 0.0% | 223 | 0.0% | 376 | 0.1% | 52,958 | 5.6% |
| Coke | 310,643 | 8.1% | 2,284,857 | 29.9% | 885 | 0.2% | 138,822 | 14.6% |
| Chemistry | 870,345 | 22.8% | 552,092 | 7.2% | 12,938 | 2.8% | 188,301 | 19.8% |
| Plastic & rubber | 58,575 | 1.5% | 205,197 | 2.7% | 3,272 | 0.7% | 194,970 | 20.5% |
| Other nonmetallic | 494,812 | 13.0% | 428,399 | 5.6% | 5,588 | 1.2% | 9,842 | 1.0% |
| Metallurgy | 9,507 | 0.2% | 85,055 | 1.1% | 882 | 0.2% | 76,977 | 8.1% |
| Iron and steel | 274,826 | 7.2% | 579,842 | 7.6% | 4,335 | 0.9% | 24,024 | 2.5% |
| Car industry | 18,674 | 0.5% | 55,455 | 0.7% | 776 | 0.2% | 129,097 | 13.6% |
| Electricity prod. | 1,065,235 | 27.9% | 1,945,922 | 25.5% | 73,083 | 15.7% | 2,119 | 0.2% |
| Heat prod. | 219,328 | 5.7% | 667,978 | 8.8% | 58,162 | 12.5% | 2,004 | 0.2% |
| Gas prod. | 44,733 | 1.2% | 1,014 | 0.0% | 486 | 0.1% | 1,090 | 0.1% |
| Waste management | 184,986 | 4.8% | 110,186 | 1.4% | 290,086 | 62.4% | 6,632 | 0.7% |
| Other industries | 32,644 | 0.9% | 46,211 | 0.6% | 1,485 | 0.3% | 32,502 | 3.4% |
| Total | 3,820,350 | 100.0% | 7,632,649 | 100.0% | 464,864 | 100.0% | 950,138 | 100.0% |

of francs (1990 MF). The number of projects financed globally increased during the period, reaching a maximum of 63 in 1997. The total amount distributed was quite high in 1991 and decreased regularly until 1996. During the last two years, subsidies almost doubled (160MF in 1997 and 153MF in 1998).

These subsidies are not distributed equally between sectors as shown in Table 2. The chemistry sector got the highest number of abatement projects financed (70 projects), followed by the sector of coke (36), plastic and rubber (30) and waste management (28). We note also that the sector of gas production and the sector of extraction did not get any subsidy between 1990 and 1998. In terms of monetary amount, 40% of total subsidies were equally shared between the sectors of chemistry and electricity production (210MF each). The sectors of coke and of iron and steel received respectively 13% and 11% of total subsidies. It is no surprise that subsidies are granted mainly to the biggest polluters.

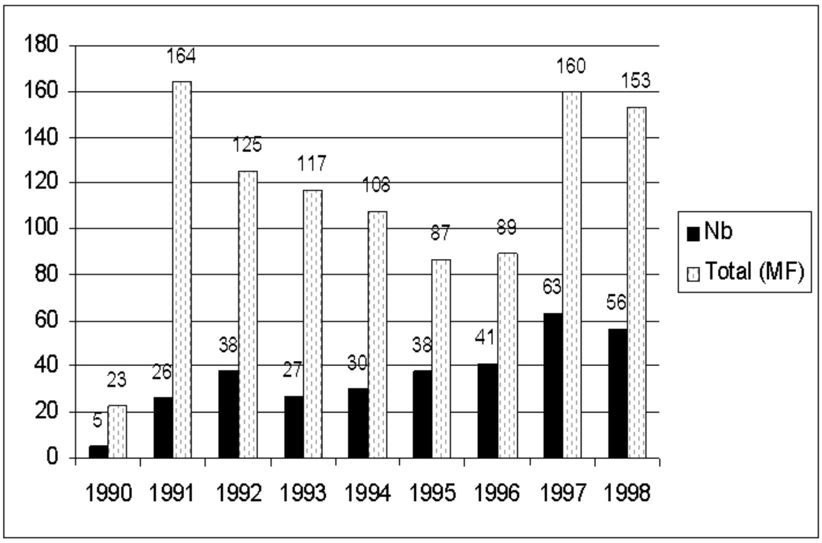


Figure 2: Number and total amount of subsidies for each year

Table 2: Number and total amount of subsidies by sector for the 1990-1999 period

| Sector | Nb | Tot. (MF) | % |
|---|---|---|---|
| Extraction | 0 | 0 | 0% |
| Food & agri. prod. | 5 | 7 | 1% |
| Textile | 6 | 7 | 1% |
| Paper | 14 | 36 | 4% |
| Printing | 14 | 23 | 2% |
| Coke | 36 | 134 | 13% |
| Chemistry | 70 | 208 | 20% |
| Plastic & rubber | 30 | 52 | 5% |
| Other nonmetallic | 21 | 40 | 4% |
| Metallurgy | 15 | 18 | 2% |
| Iron and steel | 24 | 114 | 11% |
| Car industry | 11 | 62 | 6% |
| Electricity prod. | 13 | 210 | 20% |
| Heat prod. | 11 | 10 | 1% |
| Gas prod. | 0 | 0 | 0% |
| Waste | 28 | 66 | 6% |
| Other industries | 26 | 38 | 4% |
| Total | 324 | 1,025 | 100% |

# 6 Theoretical model

For the theoretical framework we use the model of Khanna and Zilberman (1997) who formalize the view of pollution as a result from waste in input use. Consider a plant using a polluting input in its production process (for example, energy from fossil fuel, or solvant-based paints). To simplify the analysis, assume that the plant produces a single output, $q$, and that the plant is a price-taker both in the input and output markets. A plant can choose to invest ($i = 1$) or not ($i = 0$) in a new equipment that would increase input-use efficiency. The production function $f$ is written as a function of useful input $e$ and a productivity parameter $\beta$ : $q_i = f(\beta_i e_i)$ and possesses the usual properties: $f_e > 0$ and $f_{ee} < 0$. The parameter $h$ is used to account for efficiency in input use, where $h$ is the ratio of useful input to applied input: $h_i(\theta) = \frac{e_i}{a_i}$ . The production function can thus be written $q_i = f(\beta_i h_i(\theta) a_i)$. Firms are heterogenous in that the input use efficiency depends on management or other firm characteristics: $h(\theta)$ with $\theta$ distributed on a scale from 0 to 1. Input use efficiency increases with the heterogeneity index, but at a decreasing rate: $h_\theta > 0$ and $h_{\theta\theta} < 0$. Adopting the input-saving technology improves efficiency: $h_1(\theta) > h_0(\theta)$ for $0 < \theta < 1$. In other words, less applied input is required with the new technology for producing the same level of output (or equivalently the production function with the new equipment is always above the production function without the new equipment). Whereas efficiency in input use depends on the firm type and on technology, the parameter $\beta$ is a direct productivity effect associated only with the technology itself. For all $\theta$, a precision technology is defined as a technology with $\beta_1 > \beta_0 = 1$. Investing in the new technology implies a fixed cost ($I_1 > 0$ and $I_0 = 0$) and increases the marginal cost of input ($w_1 > w_0$) (to account for a more skill-intensive technology). $P$ is the unit output price. The cost function is written $C_i(a) = I_i + w_i a_i$. The profit function thus reads $\Pi_i(a_i) = Pf(\beta_i h_i(\theta) a_i) - C_i(a_i)$.

## 6.1 Benchmark case

As a benchmark, define the situation without any regulation at all. In a first stage, the plant chooses the optimal level of input with and without the new equipment solving the program:

$$Max\ \Pi_i(a_i) = Pf(\beta_i h_i(\theta) a_i) - C_i(a_i); \ \ i = 0, 1.$$

Optimal input use $a_i^*$ is determined by the following first-order condition:

$$P\beta_i h_i(\theta) f'(\beta_i h_i(\theta) a_i^*) = w_i; \quad i = 0, 1. \tag{1}$$

The value of the marginal product of input should equal its marginal cost. When $i = 1$, we have $\beta_1 f'(\beta_1 h_1(\theta) a_1^*) = w_1/(Ph_1(\theta))$ and when $i = 0$, the optimal input is given by $f'(\beta_0 h_0(\theta) a_0^*) = w_0/(Ph_0(\theta))$. Whenever the marginal productivity effect of the new technology exceeds its marginal cost increase, applied input use is higher with the new technology than with the old: $a_1^* > a_0^*$ when $\beta_1 \frac{h_1}{h_0} > \frac{w_1}{w_0}$.

In a second stage, the plant compares the profit obtained with and without the new equipment. New equipment will be acquired if $\Pi_1^* > \Pi_0^*$, or equivalently if

$$I_1 < P[f(\beta_1 h_1(\theta) a_1^*) - f(h_0(\theta) a_0^*)] + w_0 a_0^* - w_1 a_1^*. \tag{2}$$

A plant is more likely to adopt the new equipment if the fixed investment cost $I_1$ is low, the revenue differential $P(q_1^* - q_0^*)$ (which is positive if $a_1^* > a_0^*$) is high and the cost differential $(w_1 a_1^* - w_0 a_0^*)$ is low.

### 6.2 Introducing emissions regulation

Assume now that input consumption produces a proportional amount of pollution: the total amount of emissions $z$ is a constant share $\gamma$ of the applied input. Equivalently, we have the relationship $z_i = \gamma_i a_i$. All else equal, the adoption of a new precision technology reduces the pollution coefficient and $\gamma_1 < \gamma_0$. Existing French regulation can be modelled as an emission standard $N$ backed up by a penalty for non-compliance $F$. Assuming that the standard is binding, the firm maximizes

$$\Pi_i(a_i) = Pf(\beta_i h_i(\theta) a_i) - C_i(a_i) - F(\gamma_i a_i - N)\,; \quad i = 0, 1.$$

and chooses a level of input use equal to

$$\beta_i f'(\beta_i h_i(\theta) a_i^*) = \frac{w_i + F'\gamma_i}{Ph_i(\theta)}; \quad i = 0, 1. \tag{3}$$

As expected from theory, with perfect enforcement the result resembles an emission tax. An emission tax increases the marginal cost of applied input, leading to a reduced level of output (comparing equation 3 with equation 1). However, if enforcement is less than perfect and exceptions and delays are accorded, combining existing regulation with an emission tax $t$ would give additional incentives for polluting firms. In the French regulatory situation with the TPPA, the firm thus maximizes

$$\Pi_i(a_i) = Pf(\beta_i h_i(\theta)a_i) - C_i(a_i) - F(\gamma_i a_i - N) - t\gamma_i a_i; \;\; i = 0, 1.$$

Optimal input use is now given by

$$\beta_i f'(\beta_i h_i(\theta)a_i^*) = \frac{w_i + F'\gamma_i + t\gamma_i}{Ph_i(\theta)}; \;\; i = 0, 1. \tag{4}$$

In both cases, the optimal applied input $a_i^*$ is lower than in the scenario without a tax. Now, a plant will invest if

$$I_1 < P[f(\beta_1 h_1(\theta)a_1^*) - f(h_0(\theta)a_0^*)] + w_0 a_0^* - w_1 a_1^* + F(\gamma_0 a_0^* - \gamma_1 a_1^*) + t\gamma_0 a_0^* - t\gamma_1 a_1^*. \tag{5}$$

Compared to the benchmark case, a plant is now more likely to invest in a clean technology the higher is the tax rate and the higher is the regulatory threat of enforcing an emission standard.

The linear relation between applied input and emissions in this model holds as a good approximation for $SO_2$ emissions and more or less for use of hydrochloric acid and solvants leading to VOC emissions. For $NO_x$ emissions, on the other hand, the link between the energy input and resulting emissions is more intricate. It depends amongst other things on the process combustion temperature. The model should thus be interpreted as a first approximation of the firm's input use and choice of adopting a clean technology. Below, we present preliminary results from estimation of Equation (4), which determines the emission level given the technology choice.

# 7 Econometric analysis

Observations come from an unbalanced panel data set. Estimations are made for the 1,915 plants with strictly positive tax payments (if the tax due did not exceed 1,000FF, no tax was levied). We estimate separately four models, one for each pollutant. Regarding $NO_x$ and $SO_2$, the dependent variable is the amount of yearly emissions by unit of combustion capacity (i.e. total emissions of the plant / maximum combustion capacity). This is to avoid size effects in the estimation. In the case of VOC and HCl, the dependent variable is the total amount of yearly emissions (we were not able to control for size for those pollutants).[13] The theoretical framework

[13] In the case of VOC, the regression is run on data posterior to 1995, when taxation of this pollutant began.

showed how emissions depend on input use, which in turn is determined by input price, output price, firm characteristics, regulatory norms and the tax level. Regulatory norms are firm-specific in France and it is impossible to obtain data on those. We include as regressors the level of the tax (tax per tonne for the pollutant considered), the sectoral dummies,[14] and the regional Gross Domestic Product (to control for economic activity in the regions). In the case of $NO_x$, $SO_2$ and VOC we also introduce a proxy for the price of oil, the one input for which prices are readily available. We use the Brent index for crude oil (source: Direction des Ressources Energétiques et Minérales in France, 2003), which varies with time only. The panel form of the data allows us to control for unobserved plant-specific effects. The four models are estimated using Generalized Least Squares. Estimation results are reported in Table 3.

Quality of adjustment varies from 0.11 (R-squared) in the model fitting HCl emissions to 0.25 in the model for $NO_x$. A large part of the emissions variability still remains unexplained by these preliminary estimations. The unit tax has the expected negative sign in all cases even if it is significant in three models over four only. Except for the case of VOC, a greater level of the unit tax leads to a significant reduced amount of pollution, all other things being equal. The elasticity of emissions with respect to the tax is -0.15 for $NO_2$, -1.40 for $SO_2$, -0.06 for VOC and -2.71 for HCl.

Many sectoral dummies are statistically significant with respect to the electricity sector, chosen as the base case. The estimation results confirm the importance of particular industries for the emissions of certain pollutants. All else equal, a firm in the coke sector or in the metallurgy sector has much higher $SO_2$ emissions per unit of combustion capacity. Likewise, a firm in the coke sector, in the car industry or in the metallurgy sector, would have significantly higher emissions of VOC.

The coefficient of the regional GDP is almost equal to zero in all four models. So we do not find any significant impact of the regional economic activity on the level of emissions. It is likely that regional GDP is too crude an indicator to control for the level of economic activity. Future research will incorporate firm-specific variables to account for the level of economic activity (value added or turnover).

The Brent index for crude oil has a significant and negative impact on $SO_2$ emissions. This is what was expected as most of $SO_2$ emissions originate from energy combustion. The estimated

[14] In each model, the electricity sector is chosen as the reference.

Table 3: Estimation results for the four pollutants

| | $NO_x$ emissions (t/MW) | | | $SO_2$ emissions (t/MW) | | |
|---|---|---|---|---|---|---|
| | Coef. | Std. Err. | $p$-value | Coef. | Std. Err. | $p$-value |
| constant | 4.3929 | 0.9625 | 0.0000 | 13.6288 | 1.3071 | 0.0000 |
| tax | -0.0024 | 0.0011 | 0.0220 | -0.0391 | 0.0026 | 0.0000 |
| Other industries | -3.2943 | 1.0665 | 0.0020 | -4.3373 | 1.4056 | 0.0020 |
| Other nonmetallic products | 1.6747 | 1.0625 | 0.1150 | 0.4621 | 1.3962 | 0.7410 |
| Plastic & rubber | -2.3496 | 1.1315 | 0.0380 | -1.7951 | 1.4781 | 0.2250 |
| Chemistry | -0.2813 | 1.0498 | 0.7890 | -0.7122 | 1.3922 | 0.6090 |
| Coke | -1.8098 | 1.4724 | 0.2190 | 9.0411 | 1.9019 | 0.0000 |
| Waste | 7.8915 | 1.0468 | 0.0000 | 2.0046 | 1.3736 | 0.1440 |
| Printing | -3.9758 | 1.9087 | 0.0370 | -5.3157 | 2.4703 | 0.0310 |
| Food & agri. products | -3.2601 | 1.0129 | 0.0010 | -2.7461 | 1.3337 | 0.0390 |
| Car industry | -3.5236 | 1.2303 | 0.0040 | -4.1642 | 1.6392 | 0.0110 |
| Extraction | -2.7548 | 1.7915 | 0.1240 | 0.4207 | 2.2287 | 0.8500 |
| Paper | -2.7039 | 1.1142 | 0.0150 | -0.4234 | 1.4778 | 0.7740 |
| Textile | -3.5019 | 1.2296 | 0.0040 | -2.9778 | 1.6360 | 0.0690 |
| Metallurgy | -3.4187 | 1.3786 | 0.0130 | 3.4821 | 1.8291 | 0.0570 |
| Heat production | -3.3590 | 0.9890 | 0.0010 | -3.7792 | 1.2996 | 0.0040 |
| Gas production | -0.7746 | 1.4550 | 0.5940 | -5.4588 | 2.4056 | 0.0230 |
| Iron and steel | -2.6724 | 1.1301 | 0.0180 | -3.2091 | 1.4764 | 0.0300 |
| Regional GDP | 0.0000 | 0.0000 | 0.1620 | 0.0000 | 0.0000 | 0.0010 |
| Brent index | 0.0164 | 0.0095 | 0.0840 | -0.0435 | 0.0088 | 0.0000 |
| Nb of obs. | 11,230 | | | 10,672 | | |
| $R^2$ | 0.25 | | | 0.12 | | |

| | VOC emissions (t) | | | HCl emissions (t) | | |
|---|---|---|---|---|---|---|
| | Coef. | Std. Err. | $p$-value | Coef. | Std. Err. | $p$-value |
| constant | 39.2870 | 99.2579 | 0.6920 | 717.3484 | 66.9084 | 0.0000 |
| tax | -0.0860 | 0.0703 | 0.2210 | -2.1878 | 0.2646 | 0.0000 |
| Other industries | 107.2305 | 107.9028 | 0.3200 | -317.1838 | 81.4641 | 0.0000 |
| Other nonmetallic products | 27.0947 | 110.9392 | 0.8070 | -324.0153 | 59.1940 | 0.0000 |
| Plastic & rubber | 377.1087 | 104.5526 | 0.0000 | -306.4785 | 76.7523 | 0.0000 |
| Chemistry | 361.9981 | 104.1611 | 0.0010 | -295.0163 | 64.6446 | 0.0000 |
| Coke | 746.1684 | 120.1007 | 0.0000 | -284.3979 | 151.9854 | 0.0610 |
| Waste | -5.2562 | 101.9789 | 0.9590 | -131.9511 | 55.4152 | 0.0170 |
| Printing | 395.5574 | 127.8858 | 0.0020 | 56.5190 | 302.7676 | 0.8520 |
| Food & agri. products | 161.7766 | 126.9678 | 0.2030 | -349.6873 | 61.2467 | 0.0000 |
| Car industry | 515.7755 | 115.0877 | 0.0000 | -333.3831 | 106.3163 | 0.0020 |
| Extraction | 31.4620 | 322.2374 | 0.9220 | -172.4242 | 134.0631 | 0.1980 |
| Paper | 378.9330 | 116.7600 | 0.0010 | -304.8827 | 82.8141 | 0.0000 |
| Textile | 132.2103 | 134.3641 | 0.3250 | -342.3384 | 90.0767 | 0.0000 |
| Metallurgy | 462.2797 | 123.1043 | 0.0000 | -284.9776 | 97.8360 | 0.0040 |
| Heat production | 14.8264 | 135.5255 | 0.9130 | -309.7334 | 57.2706 | 0.0000 |
| Gas production | 118.5143 | 244.3531 | 0.6280 | 121.7373 | 302.5430 | 0.6870 |
| Iron and steel | 119.0981 | 118.8534 | 0.3160 | -307.8316 | 74.0657 | 0.0000 |
| Regional GDP | 0.0000 | 0.0002 | 0.8690 | 0.0000 | 0.0001 | 0.8490 |
| Brent index | -0.4268 | 0.7282 | 0.5580 | . | . | . |
| Nb of obs. | 3,061 | | | 3,320 | | |
| $R^2$ | 0.20 | | | 0.11 | | |

* $NO_x$ include both $NO_2$ and $N_2O$. However, we exclude $N_2O$ from the analysis because the union of chemical producers has been refusing to pay the tax for this pollutant since 1990.

coefficient is also negative in the model for VOC but it is not significant. The positive sign found in the $NO_x$ model is somewhat surprising.

Overall, the estimations show that the tax had a significant negative effect, but that the abatement elasticity with regard to the tax rate was quite small. Furthermore, these estimations provide an upper bound on the abatement elasticity. Environmental taxes in Europe seldom operate in isolation, but are rather to be seen as a complement to traditional command-and-control regulation. A shortcoming of the current work is that we have not been able to control for the technology standards defined in operating permits for each industry. Such standards have most likely not changed over the period, however, and some of the inter-firm variation in standards may have been captured by the industrial sector dummies in the current analysis.

# 8 Conclusions

We have estimated a random-effect model for air pollution emissions on a panel data set comprising around 1,900 French firms over the period 1990-99. Despite the relatively low level of the tax, statistically significant reductions in emissions are found for emissions of $SO_2$, $NO_x$ and HCl. The estimations show that the instrument has had the largest impact on $SO_2$ and HCl emissions. On the other hand, the overall effectiveness and economic efficiency of the French air pollution tax can only be assessed by a complete model incorporating subsidies as well as taxes. The next step in the research is therefore to include the effect of the subsidies paid out of the revenues from the tax, which in fact was earmarked for abatement subsidies.

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