ENVIRONMENTAL POLICY: THEORETICAL FOUNDATIONS AND PRACTICAL APPLICATIONS

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1 Introduction

The theory and the practice of environmental policy is an issue that has made tremendous strides in recent decades, both at the level of research and at the level of applied policy-making. In the domain of environmental policy I would like to include policies dealing with issues such as pollution control, exhaustible or renewable resource management, ecosystem management. The possible interactions of these policies, not only among themselves, but also with other branches of economic policy such as agricultural policy, industrial policy, technology policy, and trade policy, define a much richer environment for economic analysis and also a very complex environment for practical applications. From the vantage point of economic theory, it is well known that environmental policy is a case of analyzing externalities and market failures. William Baumol and Wallace Oates (1988, p.1) in their book *The Theory of Environmental Policy,* one of the most influential works in the field, note:

"When the "environmental revolution" arrived in the 1960s, economists were ready and waiting. The economic literature contained an apparently coherent view of the pollution problem together with a compelling set of implications for public policy."

Although a framework - at least in theory - for the design of environmental policy existed, it was not the economic instruments that were used first for applied policy purposes but rather command and control type policies that did not take into account economic considerations.[1] The recent experience however shows an increasing use of economic instruments - or, as they are commonly called, economic incentives or market based instruments - in practice. These instruments tend to substitute or complement command and control issues (OECD 1989, 1991, 1994a,b, 1996a,b).

The instruments that developed as a result of the attempt to enrich the methods for designing environmental policy, the so-called market based instruments, are mainly founded on the theory of environmental policy developed in the 1970s and the 1980s. However, the continuing in-depth analysis of the nature of specific environmental problems, the enlargement of the areas of application of environmental policy, and the quest for ways to deal with important issues such as climate change, ecosystem management or agricultural run off, along with the continuous interactions of environmental economics with ecology on the one hand and other branches of economics such as international economics or industrial organization on the other, has produced a number of important extensions to the basic

[1]See Cropper and Oates (1992) for a description of the first stages of environmental policy applications in the US.

environmental theory framework. These extensions have not only helped us to better understand existing environmental problems, but have also made it possible to analyze new problems, examine the design of environmental policy in different analytical frameworks and suggest new types of policy instruments.[2]

The purpose of this paper is twofold. First it aims at presenting in a summary fashion some of the most interesting recent developments in the theory of environmental policy which have evolved in response to extensions of the basic environmental policy model, the relaxation of some of it basic assumptions, and the interactions between environmental economics and ecology and environmental economics and other branches of economics. Second it seeks to provide some information regarding the application of environmental policy in practice through market-based instruments and to evaluate, to the extent to which that is feasible, the effects and effectiveness of these instruments.

2 Theoretical Foundations

2.1 Externalities and market failure

Environmental economics has long been associated with the concepts of externalities and market failures. Although a fully satisfying definition of an externality has proven quite difficult to construct, an externality can be defined (Mas-Colell et al. 1995) as a situation where:

the well being of a consumer or the production possibility of a firm is directly affected by the actions of another agent in the economy.

As is well known, the presence of an externality, as well as the presence of a public good, is a type of market failure: a situation where some of the assumptions of the welfare theorems do not hold and as a result market equilibria do not correspond to Pareto optimal outcomes.

The presence of an environmental externality can be associated with missing markets or missing property rights. As a result the market outcome is inferior to an outcome determined by the maximization of a social welfare function, defined on environmental variables along with the other relevant choice variables. The social optimum determines the socially-optimal levels of environmental variables (emissions levels, resource stocks, desired biodiversity or ecosystem state) and provides the benchmark for comparisons with unregulated market equilibrium. These comparisons in turn determine the appropriate framework for the design of environmental policy. The main idea is to design a policy such that the regulated market

[2] For a more detailed analysis of these issues see Xepapadeas (1997).

equilibrium will produce the socially-optimal outcome for the economic and the environmental choice variables.

2.2 The Starting Point

The starting point for environmental policy design is the definitions of the unregulated market equilibrium and the social optimum. A partial equilibrium approach is usually adopted, with competitive markets, full information, no international trade, and static conditions regarding the characteristics of the pollution. Thus environmental pollution in the static model is assumed to be of the flow or fund type. For this type of pollution, the assimilating capacity of the environment is such that it does not allow the accumulation of pollutants. Thus pollution generates damages only in the period emitted and not in subsequent periods. Examples of fund or flow pollution include smoke; noise; organic pollutants that can be transformed by bacteria (in an oxygen rich environment) into substances that are not harmful; and so on (Tietenberg 1996). Considering flow type pollution allows for the use of a static analytical framework that greatly simplifies exposition.

Thus the only distortion considered in the starting model is the environmental externality. This implies that only one instrument is required to correct the distortion. As will be shown later, when more distortions are included along with the environmental externality, such as informational asymmetries or market imperfections, then more complex instruments become necessary.

2.2.1 Classical Instruments

The development of policy instruments capable of securing the socially-desirable level of environmental variables implies that an environmental regulator, or more generally a social planner, chooses levels for these variables by maximizing a criterion function. This is given by a social welfare function. The domain of the social welfare function includes environmental variables, in addition to the other relevant choice variables. The social optimum determines the optimal levels of environmental variables and provides the benchmark for comparisons with unregulated market equilibrium. The optimal environmental regulation should be chosen so that the optimal levels of environmental variables under regulation coincide with the welfare maximizing levels.

Let

$$B_i(\mathbf{e}_i) = \max_{\mathbf{x}\in X} \pi(\mathbf{x}_i, \mathbf{e}_i),\ \mathbf{x} \in X \subset \Re_+^{\ k},\ \mathbf{e} \in E \subset \Re_+^{\ m}$$

be a strictly concave reduced form function defined over environmental variables, **e**, for private profit-maximizing agent i =1,…, n. The environmental variables at this stage can be interpreted as emissions of different types of pollutants that create external damages and can be abated only at a cost.

Let also

$$W(\mathbf{e}_1,\ldots,\mathbf{e}_n) = \phi(B_1(\mathbf{e}_1),\ldots,B_n(\mathbf{e}_n)) - D(E_1,\ldots,E_m),\ E_j = \sum_{i=1}^{n} e_{ij},\ j = 1,\ldots,m$$

$$\phi' > 0,\ D \text{ strictly convex}$$

be a social welfare function. In this function $\phi(\bullet)$ can be associated with benefits from emissions, while $D(\bullet)$ can be associated with external damages due to emissions. Assume that $r(e_{ij})$ is a regulatory instrument associated with the jth environmental variable of the ith agent. The optimal regulatory instrument can be characterized as the solution of the program:

$$\max_{\mathbf{e},\mathbf{r}(e_{ij})} \phi(B_1(\mathbf{e}_1),\ldots,B_n(\mathbf{e}_n)) - D(E_1,\ldots,E_m)$$

$$\text{s. t.} \qquad\qquad\qquad\qquad \text{(OR)}$$

$$\mathbf{e}_i = \arg\max_{\mathbf{e}_i} B_i(\mathbf{e}_i) - \sum_{j=1}^{m} r(e_{ij})\ \forall i$$

If we adopt a linear function for the regulatory instrument $r(e_{ij}) = \tau_j e_{ij}$ and replace the maximizing constraints by the corresponding first-order conditions, the optimal regulation is characterized by:

$$\tau_j = \frac{\partial D}{\partial E_j}$$

If we interpret τ_j as a **Pigouvian tax** on emissions, then the above relationship implies that the optimal emission tax should be set at a level equal to the marginal damages associated with one extra unit of pollutant j.

This result can be stated as the principle: *The optimal emission tax equals marginal external damages.* This principle lies behind the structure and the design of the classical environmental policy instruments. Environmental policy instruments can be divided into two broad categories: economic incentives or market-based instruments, and direct regulation or command and control approaches. Following the OECD (1991, 1994b) classification, economic instruments include:

- environmental or emission charges or taxes
- subsidies for emission reductions
- marketable or tradable emission permits
- output taxes, deposit fund systems
- performance bonds

- voluntary agreements

On the other hand, command and control approaches include the use of limits on output, inputs, emissions or technology at the firm level. The polluting firms are required to set outputs, inputs or emissions at some prespecified level, or they are required not to exceed (or fall short of) certain predefined levels (Baumol and Oates 1988). This form of direct regulation is popular among decision-makers; however, since the early 1980s, economic instruments - which have been advocated by economists for a number of decades - have started gaining popularity in the management of environmental pollution.[3]

The instruments above constitute the basis for the application of environmental policy, as will be shown in the second part of this paper.

2.3 Departures and extensions

The definition of the instruments in the manner described above, and the main result that the linear instrument should be equal to marginal damages, require a number of restrictive assumptions.

2.3.1 Dynamics

The static analysis of environmental policy is based on the assumption that the pollutants emitted during the production activities are of the flow or fund type, and that it was the flow of emissions released into the ambient environment at any point in time that created the damages.

A very important class of pollutants, however, are those for which the stock is built into the ambient environment, as their emissions accumulate at a rate exceeding the rate at which natural processes can absorb them. For a stock pollutant, the damages are not caused by the flow of emissions per unit time but by the stock of the accumulated pollutants. In fact stock pollutants are associated with a number of very important environmental problems.

The anthropogenic emissions of the so-called greenhouse gases (GHGs) - carbon dioxide, chlorofluorocarbons (CFCs), methane, nitrous oxides, and ozone - resulting from the burning of fossil fuels increase the stock of carbon, as well as of the other gases, in the atmosphere. The greenhouse problem is a very good example of a stock externality since it is

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[3]OECD has played an important role in advocating the use of economic instruments for environmental management (OECD 1989, 1991). For surveys and assessment of the use of economic instruments for environmental protection, see OECD (1989, 1994a,b). For a brief history of the evolution of economic instruments in policy making, see Cropper and Oates (1992). For a discussion of the reasons that direct regulation has been preferred, see Bohm and Russell (1985) and for an attempt at integrating theory and practice regarding economic incentives, see Hahn and Stavins (1992). Hahn (1989) provides a chronicle of the introduction of marketable permits and emission charges in the USA and Europe.

not the emissions of the GHGs that cause the environmental damages, but the accumulated stock of these gases in the atmosphere.

Other examples of stock externalities include the accumulation of heavy metals such as lead in the soil, the acid depositions in soil, or the uncontrollable accumulation of nondegradeable waste in landfills. In all these cases it is the accumulation of the pollutant that creates the environmental damages.

Once, however, the notion of the stock externality is introduced, time is also explicitly introduced into the analysis. Environmental pollution becomes a dynamic process of accumulated emissions generated by production or consumption activities, and depletion of the pollutants either by natural processes, reflecting the environment's self-cleaning or assimilating capacity,[4] or by anthropogenic abatement processes.[5]

Pollution accumulation can be described as:

$$\frac{dS(t)}{dt} \equiv \dot{S}(t) = e(t) - f(S(t)) \, , S(0) = S_o \geq 0$$

where $S(t)$ is pollution accumulated at time t, S_o is the initial pollution level, $e(t)$ denotes emissions per unit time and $f(S(t))$ reflects the removal or the decay of the pollution by natural sources. In the great majority of models the removal function is assumed to reflect a constant pollution decay rate and S_o is some initial accumulation of the pollutant, so that $f(S(t)) = bS(t)$ where b is a constant exponential decay rate.[6] The social planner's problem under stock pollution becomes then,

$$\max_{\{e_1(t),\dots,e_n(t)\}} \int_0^{+\infty} \mathrm{e}^{-rt} \left[\sum_{i=1}^{n} B_i(e_i(t)) - D(S(t)) \right] dt$$

$$\text{s.t. } \dot{S} = \sum_{i=1}^{n} e_i - bS \, , S(0) = S_o \geq 0$$

$$e_i(t) \geq 0 \, , \forall i, t$$

where r is the discount rate. The problem is usually solved by using the maximum principle,

[4]For example a large part of carbon dioxide emissions is removed from the atmosphere and is absorbed into the oceans.

[5]Dynamic models of pollution accumulation date back almost 25 years. See, for example, Forster (1973), Mäler (1974) and Brock (1977).

[6]This assumption is not free of criticism. It has been suggested (Forster 1975, Dasgupta 1982, Cesar 1994, Tahvonen and Withagen 1996, Tahvonen and Salo 1996) that if pollution is sufficiently high, then the environment's self-cleaning capacity deteriorates and may eventually become zero. Thus the decay function may take an inverted U shape, in contrast to the exponential decay assumption where the decay function increases monotonically in the stock of the pollutant.

where the current value Hamiltonian function is defined as:

$$H = \sum_{i=1}^{n} B_i(e_i) - D(S) + \lambda(t)\left[\sum_{i=1}^{n} e_i - bS\right]$$

Questions about the long-run behavior of the pollutants stock are associated with:

(i) existence of solutions;

(ii) existence and number of long-run equilibria; and

(iii) stability of equilibria.

The long-run steady-state pollution level has the local saddle point property as shown in figure 1.

The issue of the choice of the discount rate for an environmental regulator or a social planner that seeks to solve the dynamic social optimization problem, has received a great deal of attention in both theoretical and applied environmental economics.[7] In general the greater the discount rate is, the less the weight given to the net benefits accruing to the future generations.

In the uncontrolled market equilibrium, each individual firm does not take into account environmental damages so it solves the static problem $\max_{e_i} B_i(e_i)$. As can easily be deduced the uncontrolled emissions are determined by ignoring the social shadow cost of pollution. Therefore the long-run level of pollution accumulation under social planning is less than the corresponding level resulting from uncontrolled markets. Policy instruments can be defined in terms of time dependent emission taxes, limits or dynamic permit systems. The basic condition for a linear time-dependent policy instrument is that

$$\tau(t) = \lambda(t)$$

The dynamic model can be extended by allowing for capital accumulation by the firms. Models can be set up which analyze the firms' response to different policy instruments in this dynamic set up (Xepapadeas 1992, Kort 1995).

2.3.1.1 Strategic Behavior in Dynamic Models

An interesting extension of these models is the introduction of stock effects into the objective function of the private agent. This can be related to resource management problems with (biomass) stock effects in the harvesting or the cost function, or with ambient pollution stock effects in production functions. In this case the private agent's problem takes the form:

[7]See for example the special issue of the *Journal of Environmental Economics and Management* (1990) and Weitzman (1994).

$$\max_{\{e_i(t)\}} \int_0^{+\infty} e^{-rt}\left[G_i(e_i, S)\right]dt$$

$$\text{s.t. } \dot{S} = e_i + \sum_{j \neq i}^{n} e_j - bS\,,\ S(0) = S_o \geq 0$$

$$e_i(t) \geq 0, \forall i, t$$

This is a differential game model. Solutions can be obtained in the form of Open Loop Nash Equilibrium or Feedback Nash Equilibrium. In general the deviation between the private and the social solution depends on the deviation between the social shadow cost of the externality $\lambda(t)$ and the private shadow cost of the externality $\lambda_i(t)$, where $\lambda(t) > \lambda_i(t)$. Policy instruments can be defined based on the difference $\lambda(t) - \lambda_i(t)$.

2.3.1.2 Non Linear Dynamics

The qualitative behavior of the dynamic model changes drastically if we introduce non-linearities into the pollution accumulation equation. Write the pollution accumulation as

$$\dot{P}(t) = \sum_{i=1}^{n} L_i(t) - sP(t) + r\frac{P(t)^2}{P(t)^2 + m^2},\ P(0) = P_0 \qquad (1)$$

which is an equation associated with accumulation of phosphorus in shallow lakes.[8] In this equation P is the amount of phosphorus in algae; L is the input of phosphorus (the "loading"); s is the rate of loss consisting of sedimentation, outflow and sequestration in other biomass; r is the maximum rate of internal loading; and m is the anoxic level (see Carpenter and Cottingham 1997; Scheffer 1997). Estimates of the parameters of the differential equation for different lakes vary considerably, however, so that a wide range of possible values has to be considered.

By substituting $x = P/m$, $a = L/r$, $b = sm/r$ and by changing the time scale to rt/m, equation (1) can be rewritten as

$$\dot{x} = a - bx + \frac{x^2}{x^2 + 1},\ x(0) = x_0\,, a = \sum_i L_i \qquad (2)$$

In order to understand the essential features of the model suppose that the loading a is constant. For high values of a, equation (2) always has one stable equilibrium. For lower values of a, three things can happen depending on the value of the parameter b. If b 33/8, all values of a lead to one stable equilibrium (figure 2.a). If b 0.5, a range of values of a exists, starting at 0, where equation (2) has one high stable equilibrium and one low stable equilibrium. The third root of the right-hand side of equation (2) determines the borderline between the two domains of attraction. If 0.5 < b < 33/8, the range with two stable equilibria is preceded by a range of low values of a with again only one stable equilibrium.

It is easy to see the hysteresis effect now for $b < 33/8$. If the loading a is gradually increased, at first the equilibrium levels of phosphorus remain low so that the lake is in an oligotrophic state with a high value of ecological services. At a certain point, however, the lake flips to a eutrophic state with high equilibrium levels of phosphorus and a low value of ecological services. If it is now decided to lower the loading a in order to try to restore the lake, it has to be decreased below this flipping point until a point is reached where the lake flips back to an oligotrophic state (Figure 2.b). If b . 0.5, the lake is trapped in high equilibrium levels of phosphorus, as can easily be seen from figure 2.c, which means that the first flip is irreversible. In that case only a disturbance of the parameter b of the model, e.g. by changing the fauna of the lake, can possibly restore the lake.

Formulating the problem in terms of optimal control of the loading could lead for a wide range of parameters to multiple steady-state equilibria. Multiple equilibria are also obtained when the problem is formulated as a differential game with open loop or feedback information structures (Figure 3a, 3b, 3c). In this case policy schemes can be defined in the form of steady-state taxes that could direct the regulated market to the desired steady state (Figures 4a, 4b).

2.3.2 Informational Asymmetries

In contrast to point source (PS) pollution problems where the source, the size and the distinctive characteristics of the discharges can be identified with sufficient accuracy at a non-prohibitive cost, in non-point source (NPS) pollution problems neither the source nor the size of the individual emissions can be observed by an environmental regulator seeking to implement a given environmental policy. Both in theoretical and applied environmental economics, PS pollution problems have traditionally been associated with large industrial or municipal emissions, while NPS pollution problems relate mostly to emissions by small sources like farmers or households, or mobile sources such as vehicles. The pollution that these sources generate mainly includes nutrient pollution, pesticide pollution, sedimentation, vehicle pollution, and hazardous and solid wastes.

In all of these cases monitoring of the individual emissions which are associated with farming or forestry activities, with acid rain, or with urban drainage, and which are responsible for environmental degradation, is not possible due to the number of sources and the diffused character of the pollution. In many cases critical pollution-generating inputs are

[8] See Brock et al. (1998), Mäler et al. (1999).

not always observable while weather conditions introduce stochastic elements into the pollution dispersion process, making identification of the polluting source and its contribution to the ambient pollution in the specific receiving body practically impossible. Thus in an NPS problem an environmental regulator can measure the ambient pollution at specific "receptor points", but can not attribute any specific portion of the pollutant concentration to a specific discharger. Therefore the problems that characterize an NPS pollution problem are mainly informational, and have been distinguished by Braden and Segerson (1993) into two broad classes: problems related to monitoring and measurement, and problems related to natural variability.[9] Monitoring problems are associated with the inability to directly observe individual emissions or to infer them from observable inputs or from the ambient concentration of the pollutant. Natural variability is associated mainly with weather conditions or technological uncertainty and results in stochastic pollution processes.

The informational asymmetries between the regulator and individual dischargers in an NPS problem could take the form of moral hazard characterized by hidden actions or/and adverse selection. Inability to observe the emissions of each potential polluter is associated with moral hazard, while inability to know the specific characteristics or type of each potential polluter - which is private information known only to the polluter - is associated with adverse selection. In a situation characterized by these informational asymmetries, the environmental regulator can not use standard instruments of environmental policy such as Pigouvian taxes, tradeable emission permits and emission standards as a means of inducing dischargers to follow socially-desirable policies. The potential dischargers will choose higher than socially-desirable emission levels if by doing so they can increase their profits, since their emissions can not be observed and the standard environmental policy instruments can not be used to internalize external damages and to obtain the Pareto optimal outcome. Policy schemes capable of dealing with this situation are summarized in the following section.

Efficient environmental regulation in an NPS pollution problem is not possible by using the conventional policy instruments derived for PS pollution problems. Direct and indirect approaches which have been developed to determine incentive schemes include:

- Ambient taxes (taxes based on the ambient concentration of a pollutant, that all potential dischargers pay, irrespective of their own unobserved discharges).
- Input-based schemes where taxes are imposed on the use of observable polluting inputs.
- Combined ambient and traditional emission taxes for the part of individual output that can

[9]For a survey of issues related to NPS pollution problems see Xepapadeas (1999b).

be observed either by some partial monitoring or by revelation by the individual dischargers in order to avoid the ambient tax.

- Collective penalties imposed on all potential dischargers once ambient pollution exceeds some desired limits.

Because of the complexity of the problem, the informational requirements for the application of the incentive schemes for NPS pollution regulation are formidable and their political feasibility, especially for some collective penalty schemes, is not always guaranteed. Policies aimed at increasing the informational basis of the regulator regarding emission observability[10] could transform an NPS pollution problem into a PS one and allow the use of more conventional and acceptable policy instruments.

2.3.3 Market Imperfections

The assumption of competitive product markets has been the most common one in the analysis of environmental policy during the last decades,[11] while some notable exceptions refer to the use of the monopoly assumption in the product market.[12] Much less attention, at least until recently,[13] has been given, however, to the analysis of environmental policy under the assumption of oligopolistic product markets, although this assumption could be regarded as the more realistic one for describing modern industrial societies.

The basic implication of the departure from the competitive market assumption is that more externalities in addition to environmental pollution enter the analysis, with the presence of these new externalities affecting - sometimes significantly - the effectiveness of the environmental policy instruments.

In particular when a second externality related to the structure of the product market is considered, this second externality affects the product side and relates to underproduction due to excessive monopoly power, as compared to the competitive case. In the case of two distortions, standard arguments suggest the use of two instruments in order to correct the two externalities: one instrument to correct for environmental pollution and another to correct for market imperfections. In most of the interesting cases, however, the regulator cannot affect the firm's pricing policies; that is, the distortion in the product side cannot be corrected. In

[10]See Xepapadeas (1994) for a model incorporating optimal investment policies in monitoring equipment along with the combined use of Pigouvian taxes and emission taxes.

[11]See for example Baumol and Oates (1988).

[12]It was Buchanan (1969) who first considered the implications of a monopolistic product market for environmental policy.

[13]See, for example, the collected volume by Carraro, Katsoulacos and Xepapadeas (1996).

this case optimal second-best environmental policy instruments should be developed.

Results regarding environmental policy instruments emerging from a multiple distortion world indicate:

- In the case of fixed numbers oligopolies, optimal emission taxes are in general below marginal damages, which is the case under perfectly competitive markets.
- In oligopolistic markets with free entry, optimal emission taxes could exceed marginal damages because a third distortion, excess number of firms, is created by free entry (Katsoulacos and Xepapadeas 1995).

In oligopolistic markets, the presence of more than one externality can produce sometimes even counterintuitive results. Given the complexity of the analysis under oligopolistic markets, it is virtually impossible for a single model to capture most of the possible effects. So most of the results are derived by using specific models that incorporate the structure necessary for the analysis of the particular problem .

For example Carraro and Soubeyran (1996a,b) explore models of an *n*-firms homogeneous product oligopoly where firms, which could be heterogeneous, compete à la Cournot. Firms are also subject to an emission tax, and market demand depends on an index of environmental quality. Four main results are obtained in the context of this model.

i. Under increased environmental taxation a firm's profits could increase or decrease in the oligopolistic market. This result can be contrasted with the perfect competition case in which an increase in emission taxes always causes a reduction in profits.

ii. Taxation effects on the industry as a whole are ambiguous. If emission taxes increase, industry profits and concentration may also increase instead of decreasing.

iii. If the effects of emissions on demand, and their feedback on the production of the firms, are taken into account then the ‘degree of competition’ in the industry may increase since strategic interactions among firms increase. Thus environmental policy may reduce the probability of collusive behavior in oligopolistic markets.

iv. When optimal environmental taxation is considered, firms may increase their market share and/or profits when the emission tax is increased towards its optimal level.

Market imperfections can be also combined with dynamic models as indicated by the development of dynamic duopoly models incorporating trade and emissions or dynamic oligopoly models with state dependent taxes (Benchekroun and van Long 1997).

When tradable emission permits are examined there are possibilities of market imperfections in the product market, the permit market or both. Results indicate that[14]

- The initial distribution of permits, unlike the competitive case, matters.
- Total industry profits may fall as a result of an emission trading program.
- A more asymmetric distribution of permits may increase industry profits when there is oligopoly in the product market and market power exists in the permit market.

More externalities enter into the picture if we consider the case of oligopolistic markets with firms investing in environmental R&D that causes a reduction of the emission tax bill with possible spillovers among firms. In this case environmental and industrial policy coordination can be obtained by policy schemes that include emission taxes or emission trading programs and subsidies to induce the optimal level of environmental R&D.

When firms undertake environmental R&D that can reduce the total emission tax bill, issues of time consistency of environmental policy appear. As shown recently[15] by comparing the welfare of an environmental policy consisting of an emission tax when firms undertake environmental R&D in the two cases of: (i) government commitment to the environmental tax, and (ii) government following time consistent policies without commitment, it possible to increase welfare, depending on the specific features of the market, by designing either time consistent policies, or committing to environmental taxes.

2.3.4 International Dimensions

When the assumption that emissions, or more generally environmental pollution, does not cross national boundaries is relaxed, we are able to analyze environmental problems that go beyond cases where pollution and its effects are concentrated in one country only, and move to cases where activities in one country create negative externalities not only in the country itself but also in other countries. Such problems include the pollution of rivers and lakes that border more than one country[16] - a transboundary pollution problem - and regional or global environmental problems, such as acid rains, ozone depletion and global warming.[17]

As noted by Chander and Tulkens (1992), from the point of view of resource allocation, the problem associated with transboundary or global pollution belongs to the theory of the voluntary provision of public goods, or more precisely ‘public bads’, since global pollution

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[14] Hahn (1984), Malueg (1989), Sartzetakis(1997), von der Fehr (1993)

[15] Petrakis and Xepapadeas (1999).

[16]More than two hundred river basins are multinational, and in more than fifty countries 75 per cent of their total water areas lie in international basins. This is in addition to ocean bodies which are common access international resources (Carraro and Siniscalco 1991).

[17]For a more detailed, yet compact, description of these problems, see Tietenberg (1996) and Hanley et al. (1997).

satisfies the basic characteristics of a public good, namely nonrivalry in consumption and nonexcludability.

The general methodological approach in dealing with these problems[18] is to: (i) determine the laissez-faire equilibrium where countries choose their emission levels without taking into account the external costs imposed on other counties; (ii) determine a cooperative equilibrium where countries determine their emissions so that a Pareto efficient outcome is obtained; (iii) establish the inefficiency of the laissez-faire or noncooperative equilibrium compared to the cooperative case; and (iv) propose a course of action that can achieve the efficient outcome, which is the global pollution level that satisfies the Pareto criterion.

This approach is similar to the one used to deal with local pollution problems, where the inefficiency of competitive markets as compared to the social welfare optimum is established and then appropriate policy is designed to secure the welfare-maximizing outcome. This similarity would imply that, in principle, the general policy framework for correcting environmental externalities developed in the previous chapters of this book can be used as a basis for designing policies capable of dealing with global pollution problems. There is, however, one important institutional difference stemming from the 'voluntary provision' aspect of the global pollution problems. When dealing with a pollution problem which is confined within the boundaries of one nation, whatever policy is chosen by the environmental regulator can be enforced (within of course the limitations imposed by the enforcement and informational constraints discussed in previous chapters), given the legal framework of the country which describes the ways in which such policies are implemented. When, however, a global environmental problem is examined, there is not a regulator *per se* vested with the power to enforce a given policy in a number of nations. This would require the existence of some supranational authority with the legal power to enforce policies on different nations.[19] In the absence of such an authority capable of imposing and enforcing the policy, the policy needs to be agreed upon. So as noted by Carraro and Siniscalco (1991), when international environmental problems are examined, the analysis should shift from the context of government intervention - the regulation approach - to the context of negotiations between nations and international policy coordination.[20]

[18]See for example Mäler (1989, 1990).

[19]The European Union can be regarded as such an authority. However in the EU, policies need to be agreed upon, and as discussions in recent years about the introduction of a European carbon tax reveal, agreement on environmental policies which could impose a financial burden on the member states in exchange for global environmental benefits is by no means guaranteed.

[20]For a summary of the issues related to international cooperation to protect the environment, see Barrett (1995).

Negotiations among nations should lead to some international agreement specifying policies, which should be adopted, by countries participating in the agreement. Thus an international agreement should refer either to the adoption by all countries of a specific policy instrument, like an international tax on emissions or some internationally applied quota system, or to the adoption by the signatory countries of the obligation to reduce domestic emissions in a uniform or a discriminatory way by following some type of national environmental policy.[21]

A major problem however with international agreements either to adopt an internationally-designed instrument or to reduce emissions through domestic policies, is the free-riding incentives which develop because of the common access character of the environmental problem and which can seriously impede the sustainability of the agreement. It might be in a country's best interest not to participate in the agreement to reduce emissions when the rest of the countries participate, since by doing so it can reduce its own cost of abating pollution and enjoy the benefits from the overall pollution reduction brought about by the cooperation of the rest of the countries. If countries have strong free-riding incentives, the agreement cannot be sustained.

This situation corresponds to the well-known prisoners' dilemma. In the context of a repeated game with an appropriate grim trigger strategy, free riding can be eliminated and cooperation to reduce emissions through an international agreement can be sustained. As Barrett (1991) notes, a trigger strategy can be recognized in the 1957 North Pacific Seal Treaty (Article 12).

Cooperation however cannot be sustained, even in the repeated game framework, if moving from the noncooperative equilibrium to cooperation creates gainers and losers. That is, if the cooperative solution is not individually rational. Under this situation of asymmetries among countries, extension of the time horizon cannot sustain cooperation to reduce emissions, and free riding incentives prevent international agreement unless further elaborations are made.

Thus this analysis suggests that international agreements among countries should be designed to be sustainable, that is, to overcome countries' incentives to cheat or defect from the agreement, when asymmetries require the use of side payments or issue linkage, in addition to trigger strategies, or when the repeated game framework cannot be regarded as

[21]As noted by Barrett (1995) the United Nations Environmental Programme lists 132 multilateral agreements adopted before 1991 and several that were adopted afterwards.

appropriate.[22] Therefore policy design mainly focuses on:

- how agreements leading to international cooperation regarding global environmental problems can be formed and sustained, and
- how some standard environmental policy instruments can be extended to international environmental problems so that both the structure of the policy scheme and the type of the agreement necessary for the implementation of the scheme are determined.

The basic question in the case of international environmental agreements is whether sovereign countries can voluntarily - since there is no authority to force them to cooperate - reach an agreement to protect the international commons by cutting down domestic emissions. Three main approaches have been developed in the literature regarding this issue.

2.3.4.1 Types of International Agreements

2.3.4.1a Coalition Formation

The first approach analyses the problem in terms of agreements of subgroups of countries, which seek to expand the agreement to reduce emissions by inducing other countries to join the agreement through self-financing welfare transfers.[23] This is the coalition formation approach where a group of countries commit to an environmental agreement and can then seek ways of expanding the coalition to include other countries.

Expansion of the coalition to nonsignatory countries can be obtained by transfers of resources to them. In order to have a credible mechanism of transfers the following conditions should be satisfied (Carraro and Siniscalco 1994):

i. Transfers must be self-financed.

ii. The stable coalition commitment must be satisfied. That is, the initial group of countries in the stable coalition which provide the transfers, must commit to cooperation.

iii. The committed countries should try to include the maximum number of other countries in the coalition.

2.3.4.1b Cost Sharing

The second approach analyses the problem in the context of a cooperative game with externalities and derives conditions under which a group of countries can agree to reduce emissions to a desired level in a cooperative way, and share total costs including abatement

[22]Carraro and Siniscalco (1993) note that for the case of global pollutant emissions such as CO_2 or CFCs, the repeated game framework is not particularly helpful since emission reductions involve substantial and irreversible investments, and increases in emissions as a form of trigger strategy will probably harm the triggering country. This, however, can be prevented by renegotiation proof strategies.

[23]See for example Barrett (1990, 1992, 1994a, 1995, 1997) and Petrakis and Xepapadeas (1996).

costs and environmental damages in such a way that every country is better off by cooperation.[24]

2.3.4.1c Issue Linkage

The third refers to issue linkage where agreement on the environmental issue is linked to agreement on another issue in such a way that the agreement on both issues is sustainable.[25]

The above framework for analyzing international environmental issues can be extended to include dynamic aspects of the global pollution problem which are analyzed mainly in a differential game set up,[26] linkage of environmental agreements with R&D cooperation or trade restriction, or coalition formation analysis with moral hazard.

2.3.5 Uncertainty – Irreversibility

The interaction between uncertainty and irreversibility is an issue that has received considerable attention in the environmental and resource economics literature as well as the literature of finance and investment. One fundamental proposition, established by Arrow and Fisher in the area of environmental economics, is that an option value exists associated with refraining from an irreversible decision now, when next period benefits or losses due to the decision are uncertain, even if the decision-maker is risk neutral. Closely associated with the above concepts is the concept of timing of the irreversible decision, and the question of whether the decision-maker should postpone action until more information is acquired in the future.

Recent approaches to the solution of this type of problem focus on the derivation of a free boundary derived by the solution of the associated Hamilton-Jacobi-Bellman (HJB) equation (e.g. Dixit and Pindyck 1994). The underlying intuition behind the free boundary concept in decision-making under uncertainty can be described in the following way. If the decision about undertaking an irreversible action depends on the value of a parameter that evolves stochastically in time, then there will be a critical value of this parameter such that it will be optimal to undertake the irreversible action when the observed value of the parameter is on the one side of the critical value, and not to undertake the irreversible action when the observed parameter value is on the other side of the critical value. The curve that determines this critical value for any point in time is the free boundary. Thus, the basic property of the boundary is that it divides a certain strategy space into two regions. Depending on the region

[24]See for example Chander and Tulkens (1992, 1994, 1995).

[25]See for example Cesar and de Zeeuw (1996) and Cesar (1994).

[26] See Mäler and de Zeeuw (1995).

of the space in which a stochastic variable is realized the decision-maker decides whether or not to undertake the irreversible action.

In the context of environmental and resource economics this type of analysis can be applied to problems of irreversible resource development when the returns from the resource in the developed or the undeveloped stage are uncertain (Zariphopoulou and Sheinkman 1999) or when the price associated with their resource output is uncertain. The free boundary is determined from the Hamilton-Jacobi-Bellman equation which under the assumptions usually employed in these models takes the general form:

$$\min\left\{\left[\rho V-\frac{1}{2}\sigma^2 p^2 V_{pp}-apV_p-pf(D)\right],-\left[V_D-c'(D)\right]\right\}=0$$

where V is the value function associated with the irreversible development of the resource D, p is a state variable evolving stochastically according to the geometric Brownian motion $dp_t = ap_t dt+\sigma p_t dz_t$, $f(D)$ is a benefit function associated with the developed resource, and $c'(D)$ are marginal costs associated with resource development. A free or exercise boundary is presented in figure 5. In region I, no development is undertaken. For any given *D,* random price fluctuations move the point (*D,p*) vertically up or down. If the point goes above the boundary in region II, then development is immediately undertaken so that the point shifts on the boundary. Thus optimal development proceeds gradually. In the terminology of Dixit and Pindyck (1994), this is a “barrier control” policy.

A similar approach can applied to the problem of the irreversible environmental R&D accumulation under price or policy uncertainty (Xepapadeas 1999a). When policy is carried out by emission permits whose price fluctuate randomly but at the same time unpredictable policy changes can cause discrete jumps in the price of permits, then permit prices can be modeled by the mixed Brownian motion/jump process.

$$dp_t = \eta p_t + \omega p_t dz_t + p_t dq$$

The HJB equation can then take the form:

$$\min\left\{\begin{array}{l}\left[\rho V-\frac{1}{2}\omega^2 p^2 V_{pp}-\eta pV_p-\pi(p,R)\right]+\lambda\left[V\left((1-\psi)p\right)-V\right],\\ -\left[V_R-c'(\Delta R)\right]\end{array}\right\}=0$$

where V is the value function associated with the irreversible R&D process and ψ is the jump of the Poisson process.

Comparison of the exercise boundaries derived by applying this approach under various assumptions regarding non-cooperative or cooperative behavior or social optimization allows the derivation of policy schemes under uncertainty and irreversibility.

2.3.6 Integrated Ecological/Economic modeling

The issue of integrating ecological and economic considerations is not an easy task. On the other hand, given the lack of realism regarding the modeling of ecological systems that is often encountered in economic models, any attempt to integrate ecological realism into model building with economic reasoning should be more than welcome.

An important issue in this context is that of biodiversity. Biodiversity issues mainly focus on biodiversity preservation, biodiversity valuation, or the relation of biodiversity to ecosystem resilience. One line of approach of biodiversity preservation in integrated ecological/economic modeling is the foundation of economic management modeling on mechanistic resource-based models of species competition developed in the recent decades by Tilman and his associates.

Species competition and the eventual fate of various species once competitive interactions have reached an equilibrium is an issue of fundamental importance to ecology. The phenomenological and descriptive approach of classical theory is based on the Lotca-Volterra competition. These equations, which have been the major descriptor of competition in the ecological literature since the 1920s (Lotca 1924, Volterra 1931), describe the interactions between species in terms of summary variables which are the competition coefficients.

In the last few decades a new approach has emerged based on the work of Tilman (1982, 1988), see also Pacala and Tilman (1994), Roughgarden (1998). This approach is based on a mechanistic resource-based model of competition between species and uses the resource requirements of the competing species to predict the outcome of species competition. As stated in Tilman (1988), the strength of the mechanistic approach when applied to species communities is that it can make explicit predictions about a wide range of patterns and processes in nature. A central feature of the resource-based model is an exclusion principle. This principle states that in the context of a multispecies competition for a limiting factor, in a patch free of disturbance, the species with the lowest resource requirement in equilibrium will competitively displace all other species. In this set up the system is driven to a monoculture and the equilibrium outcome of species competition is the survival of the species which is the superior competitor for the limiting resource, that is the species with the lowest resource requirement.

Of course there are a number of ways in which nature can produce equilibrium polycultures with one or more limiting resources. As noted in Pacala and Tilman (1994), temperature dependent growth and temperature variation in a habitat, spatial or temporal variations of resource ratios, differences in species palatabilities, and local abundance of herbivores, can all result in spatial or temporal variation of dominant competitors.

A general model of species competition in a landscape consisting of potentially heterogeneous patches of land can be set up as:[27]

$$\frac{\dot{B}_{ic}}{B_{ic}} = F_{ic}(\mathbf{B}_c, \mathbf{B}_{-c})G_{ic}(\mathbf{R}_c, d_{ic}),\; B_{ic} = B_{ic}^{\;0} > 0,\; \forall i, c$$

$$\dot{R}_{jc} = S_{jc}(\mathbf{R}_c, \mathbf{R}_{-c}) - D_{jc}(\mathbf{B}_c, \mathbf{B}_{-c}, \mathbf{R}_c, \mathbf{R}_{-c}),\; R_{ic} = R_{ic}^{\;0} > 0,\; \forall j, c$$

where $\mathbf{B}_c$ is the vector of biomasses in patch *c,* and $\mathbf{R}_c$ is the vector of limiting resources in the same patch. The $S(\bullet), D(\bullet)$ are the supply of and the demand for the resource respectively by all species using the resource. The growth of species is limited by resource availability, while species interact among themselves and compete for the limiting resources. This model can be regarded as a generalization of Tilman's mechanistic resource-based model of competition, by including direct interactions among species. The multi-species Kolmogorov model and the Lotca-Volterra model can also be derived from this model under appropriate assumptions.

This general model can be used to explore the implications of managing an ecosystem using economic objectives. In this context harvesting rules and biodiversity preservation in the long run can be analyzed and equilibrium biodiversity resulting from the application of different management rules as well as biodiversity achieved by undisturbed nature can be compared.

Two basic management problems are analyzed. The first is the privately-optimal management problem (POMP) where economic agents maximize solely profits from harvesting in their patch, subject to the evolution of the natural system, by regarding any interactions outside their patch as exogenous. The second is the socially-optimal management problem (SOMP) where a social planner maximizes utility from the whole landscape. Utility is derived both from harvesting and from species biomasses and all interactions among species and resources both within and across patches are taken into account.

General results obtained from this model indicate that, in general, equilibrium biodiversity under private management, social management or undisturbed nature is not the same. In particular under SOMP and Inada type conditions for marginal utility derived from

[27] See Brock and Xepapadeas (1998, 1999).

species biomasses, all species are preserved. On the other hand full preservation is not guaranteed under POMP, while the undisturbed nature provides in general a third different equilibrium. Under certain assumptions regarding prices and resource consumption, Tilman's model under private management tends to a monoculture, while the same model under a specific harvesting rule can lead to full biodiversity preservation under social management.

Furthermore the socially-optimal equilibrium biodiversity has a resilience property if the discount rate is sufficiently small, in the sense that no matter how the system is shocked the socially-optimal harvesting rule ensures that the system will return to the socially-optimal equilibrium biodiversity. When the discount rate is zero, then the socially-optimal harvesting rule can be interpreted as a golden biodiversity rule, since it maximizes steady-state utility. Given the divergence between the POMP and the SOMP, a policy scheme consisting of a biodiversity tax and a resource tax that can induce profit-maximizing agents to harvest in a way that can preserve the socially optimal biodiversity can be developed.

Another line of approach is the use of CAPM from finance to develop Biological CAPM (B-CAPM) models (Brock and Xepapadeas 1999). A two period B-CAPM model can be obtained by choosing a vector of land allocation, **w**, for different species to minimize the variance of the landscape value, $\mathbf{w}'\Omega\mathbf{w}$, subject to the constraint of achieving a given expected landscape value e_p. The value of the landscape is defined by taking into account both the harvest value of species and their existence values as unharvested biomasses remaining in the habitat.

The problem is defined as:

$$\begin{aligned} &\min_{\mathbf{w}\geq 0} \frac{1}{2}\mathbf{w}'\Omega\mathbf{w} \\ &s.t.\ \mathbf{wB} = e_p \\ &\quad\ \ \mathbf{w}'\mathbf{1} = 1 \end{aligned}$$

The nonnegativity constraint on the allocation weights corresponds to a no short sale constraint, and the solution of the problem can be used to defined efficient fully biodiversity preserving species portfolios. This approach can also be used to value biodiversity loss in terms of reduction in the expected landscape value, or increases in the variance of the landscape value (figure 6).

The B-CAPM model can be extended to an intertemporal B-CAPM model that includes the possibility of catastrophic events. Species biomasses follow mixed Brownian motion/jump processes as

$$\frac{dB_i}{B_i} = [f_i(\mathbf{w}) - h_i]dt - \sigma_\iota(\mathbf{w})dz_i - g_i(\mathbf{w})dq_i$$

where h_i is the harvesting rule per unit time.

Another extension of this model is the optimal allocation of land between monocultures and intercrops which suggests that there are benefits from integrating ideas from portfolio management with intercropping theory.

3. PRACTICAL APPLICATIONS

The analysis of the first part of the paper exposed mainly the theoretical foundations of environmental policy and indicated areas of current and areas of further research in environmental policy. However, environmental policy, like any other form of economic policy, should also be applied to the real cases for which it has been designed. Applications therefore constitute a part of environmental policy analysis which is equally important to theory. This is because applications not only target the remedying of environmental problems, which of course is the whole purpose of developing and applying the theory of environmental policy, but also because they indicate the limitations of theory and possible lines of new theoretical research.

In analyzing the practical applications of environmental policy, I would like to focus on policies based on economic incentives that give rise to the so-called market-based instruments. Economic incentives can be regarded as the second generation of environmental policy instruments, which came after the traditional command and control (CAC) environmental regulation. The CAC regulation essentially involves setting limits on output, inputs, emissions or technology at the firm level. The regulators then force emitters to comply with these limits.

In contrast to economic incentives, CAC regulation does not provide incentives to reduce the quantity of releases below permitted levels or to improve the quality of the releases of pollutants below permitted levels. This approach was proved to be cost-ineffective, and it also prohibited economic growth in certain areas (Atkinson and Lewis 1974, Tietenberg 1974, Atkinson and Tietenberg 1982, Krupnick et al. 1983, McGartland and Oates 1985).

Table 1 shows the cost of selected CAC regulation compared to the lowest cost of meeting the same objective using economic incentives.

Table 1: QUANTITATIVE STUDIES OF ECONOMIC INCENTIVE SAVINGS

Pollutants Controlled	Study, Year, and Source	Geographic Area	Command-and Control Approach	Ratio of CAC Cost to Least Cost
AIR				
Criteria Air Pollutants				
Hydrocarbons -	Maloney & Yandle (1984)	Domestic Dupont Plants	Uniform Percentage Reduction	4.15
Lead in Gasoline	U.S. EPA (1985)	United States	Uniform standard for lead in gasoline	savings of $225 from trading
Nitrogen Dioxide (NO_2)	Seskin et a1. (1983)	Chicago	Proposed RACT Regulations	14.4
(NO_2)	Krupnick (1986)	Baltimore	Proposed RACT Regulations	5.9
Particulates (TSP)-	Atkinson & Lewis (1974)	St. Louis	SIP Regulation	6.00
TSP	McGartland (1984)	Baltimore	SIP Regulations	4.18
TSP	Spofford (1984)	Lower Delaware Valley -	Uniform Percentage Reduction -	22.0
TSP	Oates et al. (1989)	Baltimore	Equal Proportional Treatment	4.0 at 90 $\mu g/m^2$
Reactive Organic Gases/NO_2	SCAQMD (Spring 1992)	Southern California	Best Available	1.5 in 1994

Source: EPA (1992), The United States Experience with Economic Incentives to Control Environmental Pollution

The major types of classical economic incentives used in practical applications are presented in table 2.

Table 2: TYPES OF ECONOMIC INCENTIVES

	Time Incentive Becomes Effective		
Incentive Type	Prior to Time of Pollution	At Time of or as a Direct Result of Pollution	Long after Pollution Occurred or Might Have Occurred
Payments to Government for Pollution		Pollution fees	
Deposit-refund Systems	Deposits		Refunds
Trading of Pollution Permits	Allowance Trading Systems		Credit Trading Systems
Payments from Government for Pollution Control	Subsidies for Installing Pollution Control Equipment		Tax Advantages in Return for Reduced Pollution
Payments to Damaged Parties under Liability Law			-Tort Law for Private Damages -Natural Resource Damages to Public Resources
Information on Pollution	Manufacturer Provided Warnings		Disclosure of Past Emissions

Source: EPA (1992), The United States Experience with Economic Incentives to Control Environmental Pollution

In the following sections the analysis will concentrate on environmental taxes and tradable emission permits which can be regarded as the most important applied market based instruments in Europe and North America.

3.1 Environmental Taxes

Environmental taxes or, as they are sometimes called, "ecotaxes" include (OECD 1996):

- Emissions taxes

  Tax payments directly related to the measured or estimated emissions
- Product charges (consumption taxes, input taxes, production taxes)

  Substitutes of emission taxes when emissions are not directly measurable or estimable
- Tax differentiation

  Variation of existing indirect taxes in favor of clean products
- User Charges

  Payments related to the environmental service delivered
- Tax relief

  Provisions in income taxes systems to encourage environmentally-friendly behavior

For the purpose of applying policy instruments, clear definition of the policy subject is also required. The EPA gives the following definitions for air emissions, which are necessary for the meaningful application of the policy instruments.

3.1.1 Types of Air Emissions

Actual emissions - The annual 'actual' emissions which have been estimated or calculated for a plant in the State Emission Inventory, added to those annual emissions that have been removed by the application of a control factor.

Allowable emissions - Maximum emissions for a pollutant that a plant or a source is allowed to discharge emission into the atmosphere legally.

Potential controlled emissions - Pollutant emissions while operating at the maximum design capacity, a schedule of 8760 hours per year and the design value of control efficiency equipment.

Potential uncontrolled emissions: Pollutant emissions while operating at the maximum design capacity and a schedule of 8760 hours per year.

3.1.2 Types of Air Emission Sources

Point Sources - The major point source emissions categories are power plants, industrial boilers, petroleum refineries, industrial surface coatings and chemical manufacturing industries. Point sources emissions are generated from stack emissions. Since most of the records are kept in AIRS/AFS, the point sources information is readily available for developing control strategies, tracking and implementation of the State Implementation Plans (SIP). For SIP inventory purposes, the point sources emissions cutoff is 10 tons per year for VOC and 100 tons per year for Nox and CO sources. For VOC sources emitting 10 tons per year or more, base year inventory emissions must be determined by each facility.

Area sources - Area sources are those emissions that are too small to be treated as point sources. Area sources emissions can be generated from solvents used for surface coating operation, degreasing, graphic arts, dry cleaning and gasoline station (tank truck unloading and refueling). Area sources are the activities where aggregated source emissions information is maintained for the entire source category rather than for each point source, and are reported at the county level.

Mobile Sources - Mobile sources are categorized for highway and off-highway sources. The highway sources include the automobiles, buses, trucks and other vehicles traveling on local and highway roads. The emissions from highway vehicles represent one-third of the overall national volatile organic compounds (VOC) and 40 percent of the overall nitric oxide (NO_x) emissions. Highway emissions are calculated using MOBILE models. States must present highway mobile source emissions by pollutant (VOC, NO_x and CO) and by individual non-attainment county. Off-highway sources are any mobile combustion sources such as railroads; marine vessels; off-road motorcycles; snowmobiles; farm, construction, industrial and lawn/garden equipment. Emissions are determined based on a source activity variable. Activity levels for each off-highway category must be developed using EPA guidance documents.[28]

3.1.3 Environmental taxes in Practice

It should be noted that *environmental taxes in practice are not truly "Pigouvian" taxes*, because it would be too difficult and expensive to determine the socially-optimal level of emissions and then estimate marginal damages at the social optimum. On the other hand, many indirect taxes - especially those associated with motor fuels; energy products; vehicles; goods like lubricants, tires; CFCs and/or halons; air transport; water use and water disposal - can be regarded as environmentally-related taxes. The following table shows taxes/excises with environmental implications, as a percent of total tax revenue and GDP in 1993.

Table 3: TAXES/EXCISES WITH ENVIRONMENTAL IMPLICATIONS

(as a percent of total tax revenue and GDP)

Country	% of total tax revenue	% of GDP
Denmark	7.30	3.65
Finland	5.40	2.47
Netherlands	6.12	2.94
Norway	10.75	4.92
Sweden	6.34	3.17

Source: OECD (1996), Environmental Taxes in OECD Countries

Table 4 shows air pollutant taxes in European countries.

Table 4: AIR POLLUTANT TAXES IN EUROPEAN COUNTRIES

	Tax Rate	Revenue Use	Introduction Year
Sulphur Dioxide			
Finland	25 ECU/kl	General budget	July 1993
France	27.2 ECU/ton	75%pollution reduction investment; 25% research and measurements	1985
Denmark	1330 ECU/ton	General budget	1996
Norway	2.1 ECU/kg	General budget	1990
Italy	53ECU/ton	Projects aimed to reduce Environmental impacts	January 1998
Spain	29.9; 32.9 ECU/ton	Revenues are earmarked	December 1995
Switzerland	7.53 ECU/ton	Contribution to the medical health insurance	January 1999
Sweden	3.41 ECU/kg S	General budget	1991
Nitrogen Oxide			
France	22.7 ECU/ton	75%pollution reduction investment; 25% research and measurements	1990
Sweden	4.SSECU/kg	Revenue is earmarked	1992
Italy	104 ECU/ton	Projects aimed to reduce environmental impacts	January 1998
Carbon Dioxide			
Sweden	120.2 ECU/kl; 124.8 ECU/ton	General budget, earmarked and recycled back to the taxpayers	1991
Denmark	14.6 ECU/ton	General budget	May1992
Norway	107.9ECU/kl	General budget	1991
Netherlands	12.4 ECU/kl; 14.9 ECU/ton	General budget	1990
Finland	36.4 ECU/kl	General budget	1990

Source: The Eco-Tax Database**,** Forum for the Future, Keele University, 1998

The SO_2 tax in France can be regarded as a true emission tax since, in contrast to the taxes imposed in other countries, it is not based on the specific sulphur content of fuels, but on the emissions directly measured or declared. As there is no direct measurement, the emitter must explain to a state-approved inspector the basis used to calculate the declared

[28] It should be noted that area sources and mobile sources correspond to non point source pollution problems for which the classical policy instruments are not appropriate.

emission level. The inspector assesses the method and outcome and if necessary corrects the taxable emission level group of the emitters. The actual tax rate is 180 FF per ton. Subject to the tax are power stations having a total capacity of over 20 MW, incinerating plants, production plants emitting more than 150 ton of sulphur compound per annum. In the latter case the marginal emission costs at the threshold are high because exceeding it induces taxation of the total emissions. The French concept aims less at the incentive impact of taxation - since the tax rate is too low - than at the beneficial effects of the subsidies financed by this tax (Cansier and Krumm 1997). Revenue from air pollution taxes was 25.2 million ECU in 1994 and 33 million ECU in 1996. Apart from some support from general research on emission measurement technology, which is around 25% of revenue, most of the tax revenues (around 75%) is earmarked for the tax-paying group (subsidies on emission abatement investments, grants for development of emission reduction and measurement technologies). Revenues are collected by the French Agency for Environment and Energy Management (ADEME). A special environmental protection committee decides how to distribute the repayment to the emitters (Speck 1998).

A question arising from the analysis of environmental taxes is whether they have any significant impact on emission reduction. Assuming a Cobb-Douglas short-run cost function where emission reduction increases costs:

$$c\left(q, e, w, \overline{K}\right) = Aq^{b_1} e^{b_2} w^{a} \overline{K}^{\gamma}, \; c_e < 0, \; c_{ee} > 0$$

and an emission tax, τ, exists, then by taking the first-order conditions for profit maximization of a competitive firm facing a price p, the short-run emission function is obtained as:

$$e = B\tau^{\frac{b_1 - 1}{1 - b_1 - b_2}} p^{\frac{b_1}{1 - b_1 - b_2}} w^{\frac{a}{1 - b_1 - b_2}} \overline{K}^{\frac{\gamma}{1 - b_1 - b_2}}$$

The effectiveness of emission taxes can be tested by examining the statistical significance of the parameter $\dfrac{b_1 - 1}{1 - b_1 - b_2}$. This test is the objective of some current research using panel data covering the period 1990-1997 for manufacturing sectors in 2-digits ISIC Classification; that is, for sectors coded from 3100 to 3900, and for the countries: Belgium, Denmark, Finland, France, Germany, Italy, Netherlands, Norway, Sweden, and United Kingdom. The tax being tested is the SO_2 tax. Table 5 shows some preliminary results.

TABLE 5: Results from fixed effects model

Industry	Tax	Price	Wage	Capital
3000	-0.002962	0.222846	0.806325	-0.71021
	(-1.5883)	(1.1198)	(5.9963)	(-1.9187)
3100	0.000066	-0.315678	0.24133	-0.227677
	(0.0525)	(-3.6912)	(1.8362)	(-1.3846)
3200	-0.002622	-0.507220	0.552261	-0.132139
	(-1.1331)	(-3.2040)	(4.3679)	(-0.7893)
3300	0.001745	0.177666	0.781373	-0.029358
	(0.4587)	(1.2081)	(5.4438)	(-0.163986)
3400	-0.003379	0.229271	0.284852	-0.205893
	(-1.6150)	(2.4817)	(1.4881)	(-1.Óâ74)
3500	-0.004359	0.293032	0.345010	-0.513940
	(-2.8211)	(2.3991)	(2.5812)	(-2.1924)
3600	0.004951	-0.300372	0.461036	0.270742
	(1.0631)	(3.5545)	(3.0889)	(1.5832)
3700	-0.004390	0.056350	0.810596	-0.045082
	(-1.3100)	(0.6622)	(4.2998)	(-0.1294)
3800	-0.004765	-0.137147	1.013705	-0.073374
	(-1.729)	(0.547835)	(7.3265)	(-0.2101)
3900	0.000636	-0.621023	0.681672	-0.003638
	(0.2279)	(-3.7782)	(5.263)	(-0.0254)

t-ratios in parentheses

These preliminary results indicate that SO_2 taxation is most likely to be effective in the sectors of paper products (3400), chemical products (3500) and fabricated metals (3800).

The recent trends in environmental taxation (OECD 1996) stress the incentive effects of environmental taxes. Thus energy taxes are being restructured in order to give the correct environmental incentive. There is also an increase in the use of product charges, and energy taxation in many countries includes taxation of the transport sector and in particular vehicles.

3.2 Emissions and Quotas Trading[29]

Under these systems emitters or resource users operate under a multi-source limit on emissions, or an aggregate limit on the resource use, and trade is allowed on permits adding up to that pre specified limit.

Relative to fee or charge-based systems, trading systems generally cost polluters less since under existing practice for allocating pollution rights, the maximum cost to polluters is the pollution control cost incurred in meeting the regulations or standards. Fee or charge based systems, on the other hand, require outlays to control pollution as well as fees or charges on al1 units of pollution that are not controlled. In addition, trading systems provide more certainty regarding total quantities of pollution than do fee-based systems unless auction approaches are used. For these and other reasons, trading systems have proven more

[29] This section draws on EPA (1992), EPA (www.epa.gov) and Sartzetakis (1999).

popular in the United States.

The main attributes of trading systems include (EPA 1992):

- Scope of coverage
- Degree of government intervention
- Technical basis for the trading
- Geographic limits for the trading

Trading programs can be applied to either inter-firm (or inter-polluter trades), intra-firm trades between product lines, and intra-firm trades between locations (which will be referred to as inter-plant trades). A trading program can involve either credits or allowances.

A credit is created by a source emitting less than its allowable limit. To obtain the credit, a polluter is required to show that its actual emissions, plus or minus any traded credits, is less than its allowable limit. In a credit program, the agency or a designated authority must certify the creation of the credit as well as record trades.

In the allowance system, trading involves future pollution. Once the environmental protection agency sets an allowable limit for a source, the source can add to its allowable limit or reduce it by trading in allowances. The agency should, at a minimum, record trades, but it need not certify each and every allowance that is traded. The certification of allowances for each source takes place prior to trading and may be revised whenever a source changes its pollution control equipment.

The degree of government intervention varies according to the program. Credit programs require monitoring the creation of credits. Agency approval is also more likely when different pollutants are traded. Mass emission limits determine the total emissions over a period of time. Other programs may prescribe the rate at which emissions occur.

The type of pollutants largely determines the geographic area over which the trades are permitted. If the pollutants spread widely, like the CFCs, the geographic area is likely to be very large. On the other hands for pollutants like CO the geographic area is likely to be small.

3.2.1 Trading Systems

- Inter-firm non-approval trading involves pollution reduction credits that, once issued, can be relatively freely traded without major intervention by pollution control agencies
- Inter-firm approval trading involves pollution reduction credits that can be used or traded by a polluter with significant intervention of pollution control agencies, usually because the pollution involved does not have identical environmental impacts
- Intra-firm approval trading involves pollution credits that are tradable only within a firm,

to be used to meet firm standards.

3.2.1.1 Inter-Firm Non-Approval Trading

Acid Rain Allowance Trading - Title IV of the Clean Air Act Amendments of 1990 directs the Environmental Protection Agency to establish a program to reduce acid rain. Acid rain is the term used to describe the phenomenon associated with emissions of fossil fue1 combustion, the transport of these emissions in the atmosphere, and the deposition of their transformation products.

The acid rain program is the largest and most successful cap-and-trade program in the world, established in 1990 to reduce industrial emissions of SO_2. The Program sets a national emissions cap equal to 50% of base year (1980) SO_2 emissions, and allocates allowances in two phases to 2,000 utility units (allocations already made for 1995 to 2030).

Phase I	January 1, 1995	440 generating units at 180 electric utility plants in 21 eastern states
Phase II	January 1, 2000	Another 1,500 generating units at 470 plants across the country

An allowance authorizes a unit within a utility or industrial source to emit one ton of SO_2 during a given year or any year thereafter. At the end of each year, the unit must hold an amount of allowances at least equal to its annual emissions, i.e., a unit that emits 5,000 tons of S02 must hold at least 5,000 allowances that are usable in that year. However, regardless of how many allowances a unit holds, it is never entitled to exceed the limits set under Title I of the Act to protect public health.

Allowances are fully marketable commodities. Once allocated, allowances may be bought, sold, traded, or banked for use in future years. Allowances may not be used for compliance prior to the calendar year for which they are allocated.

Annual public, zero revenue, auction of 2.8% of the allowances, is conducted by the EPA. The auction provides a highly visible price signal and addresses potential concerns about market power and hoarding of allowances. There are two types of allowances:

1. Spot allowances, which are valid the immediate year; and
2. Advance allowances, which are valid in future years.

Active trading in SO2 allowances began prior to the start of Phase I. The first annual auction of SO_2 allowances at the Chicago Board of Trade (CBOT) was conducted on March 29, 1993. Through June of 1999, 70.8 million allowances were traded (recorded by the Allowance Tracking System), as shown below.

Within organizations	44.0 million (62%)
Between organizations	26.9 million (38%)

At the fifth annual auction on March 1997, spot allowances were auctioned at an average price of $110.36 per ton (up from $68.14 in 1996). Estimated average price for the first two quarters of 1999 is around $200 per ton. Allowance prices are shown in the graph 1.

The Program includes high quality continuous monitoring of all emissions, high penalties for non-compliance including fines and forfeiture of allowances, and self-reporting of both actual emissions and trading activities to a public database. Full compliance was achieved.

<u>3.2.1.2 Inter-Firm Approval Trading</u>

Under these programs credits or allowances are less freely tradable. Permission of one or more governmental agencies is needed for each trade. To secure necessary permission, environmental modeling may be required and trade ratios other than unity may be set by governmental agencies. Trades may be restricted in time as well as in space.

3.2.1.2a Trading of Air Emissions Rights

The air emissions trading program consists of four separate programs.

i) Bubbles

The bubble program, first established in 1979, allows existing sources flexibility in meeting required emission limits by treating multiple emission points as if they face a single, aggregated emission limit. The bubble can include more than one facility owned by a firm or facilities owned by different firms, but al1 the affected emission points must be within the same attainment or non-attainment area.

ii) Offsets

The offset program was developed in 1976 to lessen the conflict between economic growth and progress towards air quality goals in areas that did not meet the EPA's ambient air quality standards, referred to as nonattainment areas. Without the offset policy, there was little or no opportunity to locate a major new plant or expand significantly a major existing plant ("major" generally was defined as plants emitting over 100 tons per year of one or more criteria pollutants) in areas that did not meet air quality standards. Under the offset policy, major new or modified existing sources are allowed to operate in nonattainment areas, provided that they obtain offsetting emission reduction credits from existing sources. States implementing this policy have usually required new or modified existing sources to offset emissions by a factor greater than one. Under the 1990 Clean Air Act Amendments, higher offset ratios are mandated in ozone nonattainment areas.

iii) Banking

The EPA's initial offset policy did not allow the banking of emission reduction credits for future use or sale. The offset provisions of the 1977 Clean Air Act added banking.

iv) Netting

Netting, which dates from 1980, allows sources undergoing modification to avoid new source review if they can show that plant-wide emissions do not increase significantly. Netting is the most widely used emission trading activity.

3.2.1.2b Emissions Credit Trading Programs

According to the definition of the emission credit, sources that exceed their emission standard could get emission reduction credit (ERC) by certifying their excess pollution control through the EPA. The ERCs are made transferable either within the firm (to meet emissions standards at other discharge locations) or between firms as described above.

The ERC is the currency of this program, which can be spent or stored according to the offset, bubble, banking and netting policies. Four indicative programs are described below.

i) Lead Phase Down Program

The Program's target to achieve lead-free gasoline was implemented in a number of phases.

- The initial phase of the program did not involve credit trading. Only intra-facility averaging was allowed during this time.
- The second phase of the program targeted the lead content of leaded gasoline. Marginal cost of meeting the standards at the specified deadlines differed greatly among small and large refineries.

The program allowed refineries that reduced the lead content more than the relevant standard in a given period to earn credits that could be sold to refineries or importers that exceeded the standard. In 1985, the EPA allowed refiners to bank lead credits for use until the end of 1987.

Simple reporting and no government approval requirement minimized transaction costs, which resulted in a large volume of credits traded and banked. Overall, the lead credit program was quite successful in easing the transition to the new standards.

ii) Ozone depleting substances

Implementing the limits set under the Montreal Protocol

Signatories	Date of agreement	Main targets	Deadline
24 nations	September 1998	Restriction of halons and CFCs to 50% of 1986 levels	June 1998
59 nations	July 1990	a. complete phase out of halons	December 1999

		and CFCs b. complete phase out of carbon tetrachloride c. complete phase out of methyl chloroform	 December 2000 December 2005

The EPA chose a tradeable allowance system to achieve these targets. Under this system (ozone depletion gas permits system) allowances were distributed free of charge to the affected sources (consumers and producers). These allowances were transferable within producer and consumer categories, and across borders after obtaining EPA approval.

All information on trades is confidential. Stavins and Hahn (1993) estimate that traded permits were about 10% of the total number issued.

The tradeable allowance system was later supplemented by a special tax on CFCs and halons for revenue raising purposes mainly.

iii) Regional Clean Air Incentives Market (RECLAIM)

RECLAIM is a mandatory cap-and-trade program for stationary sources that emit four tons or more of NO_x or SO_x per year implemented by the South Coast Air Quality Management District. Initial allocations were based on reported emissions between 1989 and 1991. Each source (330 facilities by 1996) was given a declining allocation of credits (number of credits decreased by 5% to 8%) each year from 1994 to 2003. Banking is not allowed.

Monitoring and reporting has been based on the use of installed continuous emissions monitors sending data to a central computer. An active trading market for credits has developed. Through 1996, more than $20 million of credits had been traded. In 1996, average transaction prices were $154/ton for 1996 NO_x RTCs, $1,729/ton for 2010 NO_x RTCs, $124/ton for 1996 SO_x RTCs, and $2,117/ton for 2010 SO_x RTCs.

iv) Effectiveness of Air Emissions Trading Programs

According to the EPA, quantitative estimates have consistently found that air emissions trading has the potential to substantially reduce industry's cost of complying with air pollution control programs. Cost savings have been commonly estimated to be 50 percent of traditional command-and control costs, and up to 95 percent in one study. Although the general feeling is that the full potential of cost savings have not been realized yet, savings from trading under the air emissions trading program probably range from $5.5 to over $12.5 billion from 1975 to the early 1990s.

The overall impact of emissions trading on air quality is likely to have been neutral. In theory, the offset ratio of greater than 1:1 ensures some reduction in pollution; however, netting may allow small increases in pollution.

3.2.2 Fisheries License Trading Systems

New Zealand has used cap-and-trade systems since 1986 for management of its commercial fisheries. The government sets Total Allowable Commercial Catch limits (TACCs) and assigns Individual Transferable Quotas (ITQ). ITQs are permanent rights to harvest fish from a particular area, and are distributed to each commercial fishing permit holder based on the holder's historic catch levels.

Up until 1990 ITQs were expressed as a fixed tonnage of fish per year, requiring the government to buy and sell quotas to ensure $\Sigma_i \mathrm{ITQ}_i$ = TACC. Since 1990 ITQs have been expressed as a proportion of the annual TACC for each fish stock.

The ITQ system has led to heavy trading, and it is estimated that 77% of all ITQ initially allocated have changed ownership. The costs of monitoring and administering the ITQ systems are similar to those of other fisheries management programs. High penalties for non-compliance, include fines and forfeiture of vessels.

Similar systems have since been introduced in Canada, Australia, Iceland and the EU.

3.2.3 Emerging International Systems

The Kyoto Protocol allows for flexibility in how and where countries achieve their greenhouse gas emission targets. Three mechanisms, which provide alternative ways of meeting the targets beyond exclusively domestic actions, were included:

- international emissions trading (trading permits among countries with Kyoto emission commitments)
- joint implementation (earning emission reduction credits by investing in emissions-reducing projects in developed countries)
- Clean Development Mechanism (involves earning emission reduction credits by investing in emissions-reducing projects in developing countries)

The Protocol did not fully specify how the flexibility mechanisms would work in practice. International negotiations are currently underway to work out these details, which indicates the close relationship between the international environmental agreement theories and the tradable permit theory.

3.2.4 Main Design Issues

The main areas of interests and applied research in the design of permits include:

Baseline and caps - The baseline for Credit Trading is specified by the pre-existing command-and-control systems. Under allowance trading, a prespecified number of allowances are allocated to polluters without reference to pre-existing standards. Thus there is a need to determine the mass emission limits of allowance trading programs as aggregate **caps** in emissions.

Temporal issues - Should sources be allowed to borrow and bank permits freely? There is also the issue of the "depreciation" of aggregate emission caps, so that the cap is reduced in time.

Equity issues - Emission trading programs create a certain allocation of burden across individuals, industrial sectors and countries. Limiting the right to emit creates scarcity value for such rights and thus the distribution of such rights becomes very important. Transparency, consistency and information sharing are very important principles to guide burden sharing.

3.2.4.1 Methods of allocation

Free allocation by criteria

Grandfathering - Allocation based on some historical emissions or activity levels. This type of allocation makes transition from command and control systems acceptable to existing sources, since it leaves them no worse-off.

Performance standards - The allocation of permits is based on current or historical ratios, or on optimal benchmark emission factors.

Auctioning

Auctioning makes permits available to everyone on the same basis. Auctioning generates revenues, which could be:

- recycled into the economy by, for example, reducing existing taxes
- used as compensation to those bearing the burden
- used to subsidize research on environmental issues

There is also the possibility of a mixed system, where a certain proportion of permits is distributed free of charge, while the rest are auctioned.

3.3 Voluntary Agreements

Voluntary agreements are a relatively new instrument of environmental policy which, however, is receiving increasing attention both in theory and in practice. A voluntary agreement is a result of negotiations between the government or an environmental regulator on the one hand, and potential polluters on the other. Reductions of emissions are obtained through an agreement that can take the form of a contract. In the contract, the firm agrees to achieve an environmental target such as emissions reduction through changes in investment patterns, technological change or waste treatment. In exchange the firm could receive subsidies in order to change its technology.

Carraro and Siniscalco (1996), in analyzing voluntary agreements, suggest that their use can be justified mainly in cases in which the target is to obtain environmental protection through technological innovation, especially in cases where market imperfection exists, or when environmental innovation has positive spillovers.

An example of regulation through voluntary agreements is the Project XL (Blacman and Mazurec 1999) where **voluntary** site specific performance standards were set at levels more stringent than the existing regulation and emitters were given the flexibility to meet the new standards in an innovative way.

It seems that voluntary agreements that promote site-specific regulation will be a more often used instrument in the future.

3.4 Non Point Source Pollution Regulation

As already mentioned above, classical instruments did not apply to NPS problems. NPS pollution problems in practice relate mainly to water pollution due to agricultural, industrial or household activities. Actual policies against water pollution which are common in many countries (OECD 1994) include user charges for sewerage and sewage treatment, water effluent charges and charges in agriculture, along with a number of more specific policies. These are general policies that do not readily conform to the stylized characteristics of the NPS pollution instruments developed in theory; nevertheless there are features that attempt even indirectly to address the non-observability of individual emissions.

User charges for sewerage and sewage treatment when measurement of the pollution load is not possible are based on water usage which provides an indirect indicator of wastewater generation. This policy can be regarded as a type of input-based incentive scheme. In fact out of eighteen countries surveyed by the OECD (1994) in only nine countries were firms charged on the basis of metered pollution load, while on the other hand, all households were charged for water use.

Water effluent charges are mainly based on metered pollution loads, thus resembling a point source instrument more. In France according to a charge administered by "Agences de l' Eau", firms can lower their effluent bill if they can prove that their emissions are lower than those estimated by the Agence. This is a case of individual emission revelation in order to reduce payment. Thus this instrument can be regarded as having some similarities, regarding the private revelation process, to the theoretical combined ambient-effluent instrument schemes. The Dutch water pollution charge has ambient tax characteristics for households and small firms, since they pay a fixed amount independent of their actual emissions. On the other hand large firms are metered. In this case we have a transformation of an NPS problem to a PS problem through metering.

Charges in agriculture are a more profound case of input-based schemes. Charges on fertilizers as applied in many countries, are based on the N- and P- content of fertilizers which are the main contributors to NPS pollution in surface water. A number of off-farm management methods also exist for reducing P- runoff such as vegetation buffer strips, riparian zones, dredging of the lake sediment.

There are also more specific policies aimed at addressing NPS pollution problems, especially in relation to agriculture. For example, in Austria there are groundwater protection zones in which if the water quality is reduced farmers have to comply with certain management practices or change land use. In Spain there are zonal programs for reducing fertilizers, in Holland there is a "manure and ammonia policy", in England and Wales codes exist which give farmers guidance to maintain good agricultural practices, while in Ireland there is a voluntary scheme for farmers to follow a specific nutrient management plan.

One of the major problems in the design of instruments for applied NPS policy, is the inability to assess the effectiveness of these instruments, given the informational constraint, the very limited direct applications of instruments, and the implementation problems associated with the application of the "pure instruments". Given these constraints a promising area of future research in NPS policy design is the assessment and the evaluation of the effectiveness and acceptability of alternative NPS policy instruments using methods of experimental economics.

4 Concluding Remarks

The purpose of this paper is twofold. First it seeks to present some of the theoretical foundations of environmental policy and then examine how the basic model can be extended in various directions, by relaxing some main assumptions. Second it looks at how the theoretical prescriptions, regarding market-based instruments, have been followed up to now. It seems that there are many promising areas of further research, in the theory of environmental policy, from which I would like to stress the introduction of nonlinear dynamics as a more realistic representation of the underlying natural systems; market imperfections, strategic behavior and time consistency issues; the analysis of informational constraints in policy design; uncertainty and irreversibly considerations in conjunction with optimal stopping rules in environmental policy; the management of biodiversity; the use of experimental methods to assess policy instruments.

On the application level it is clear that the gradual introduction of market-based instruments relies mainly on classical instruments for point source pollution problems. It seems that issues like the efficient design and the assessment of the effectiveness and various impacts of classical instruments; the design of policies to deal with nonpoint source pollution problems; and policies related to biodiversity preservation could be areas of promising applied research.

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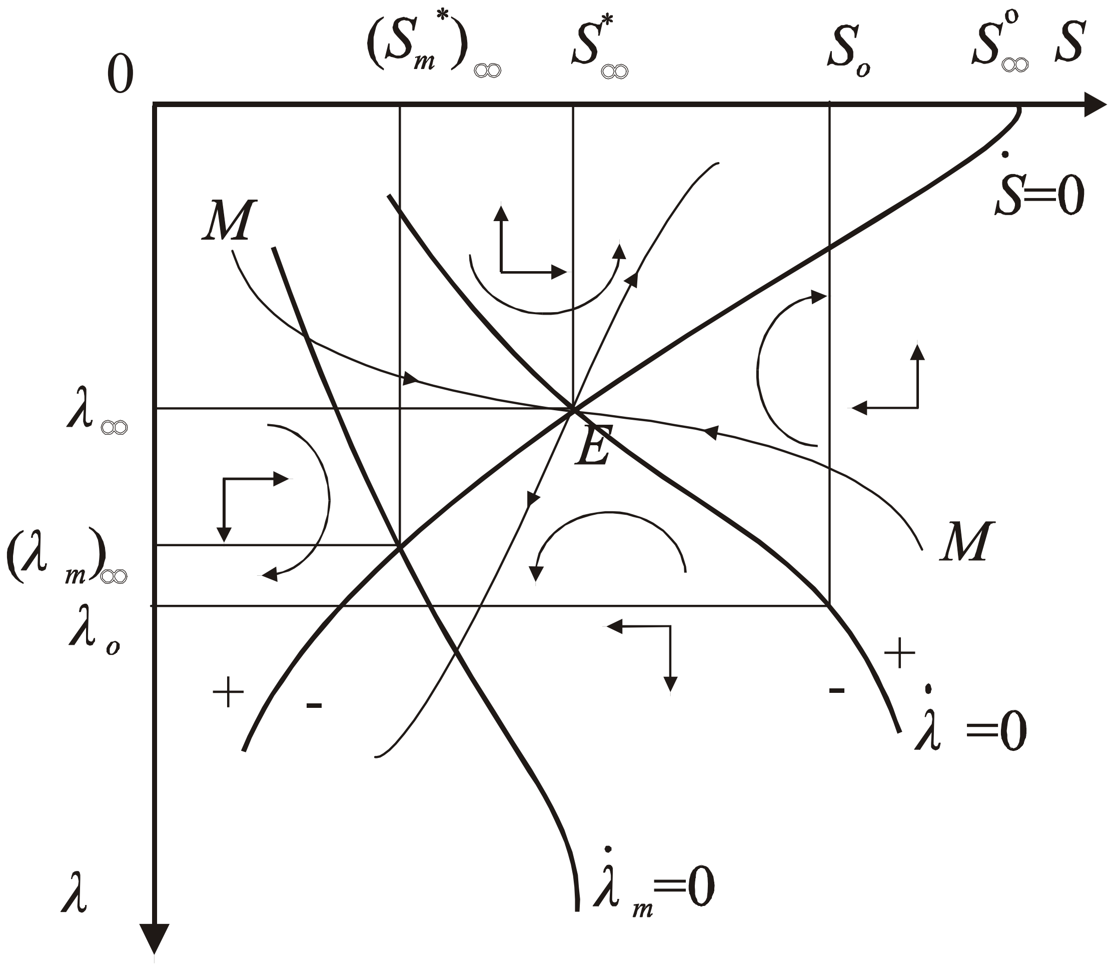


Figure 1: Long-run equilibrium for the pollution stock

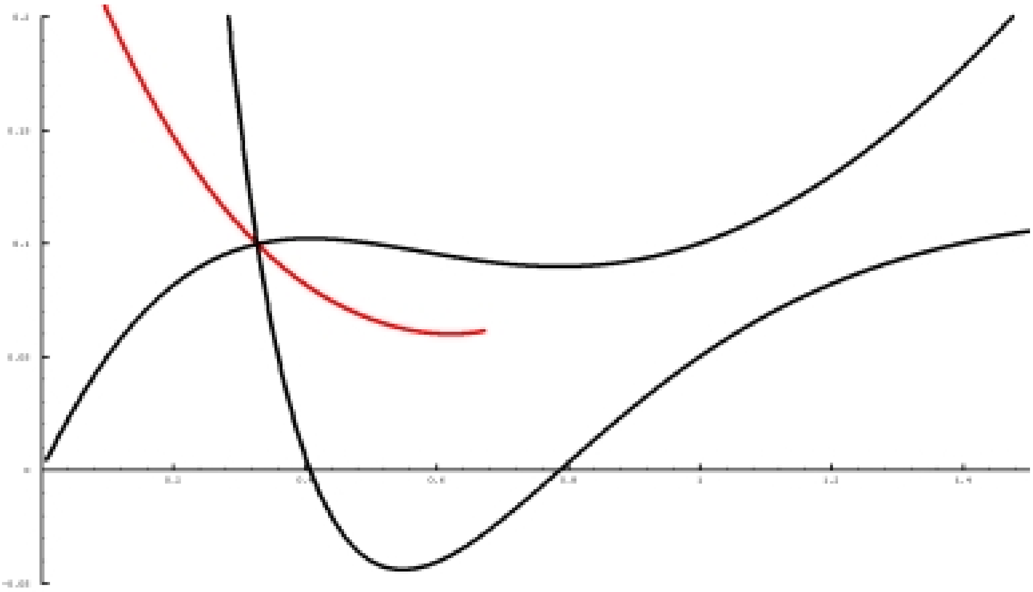


Figure 3a: Equilibrium for the planners problem

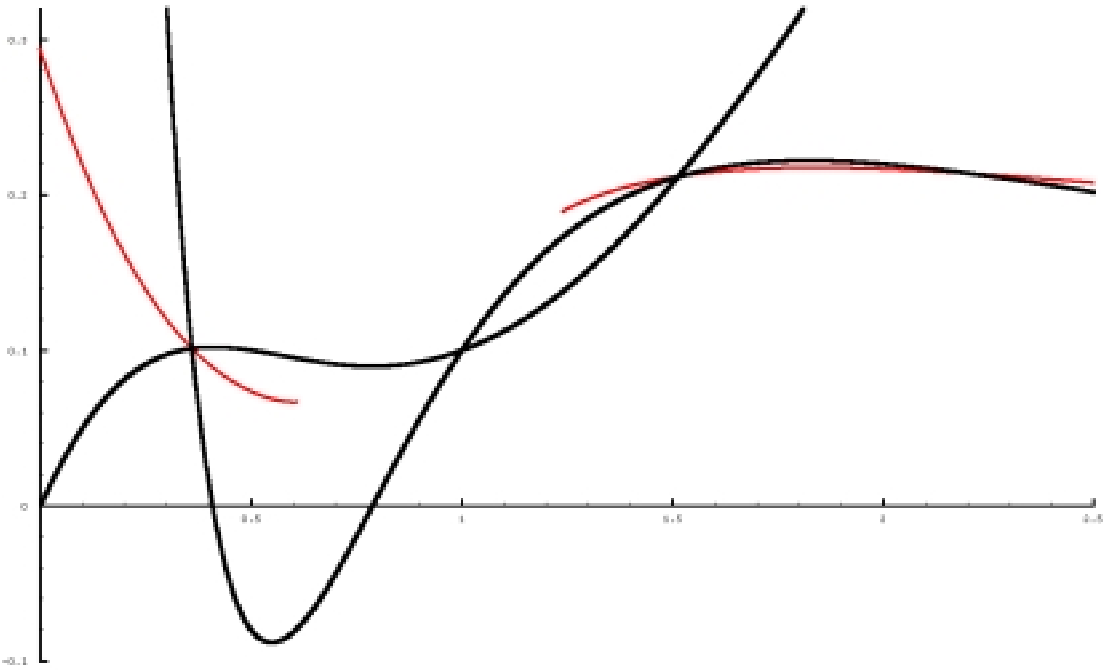


Figure 3b: Multiple equilibria under open loop information structures

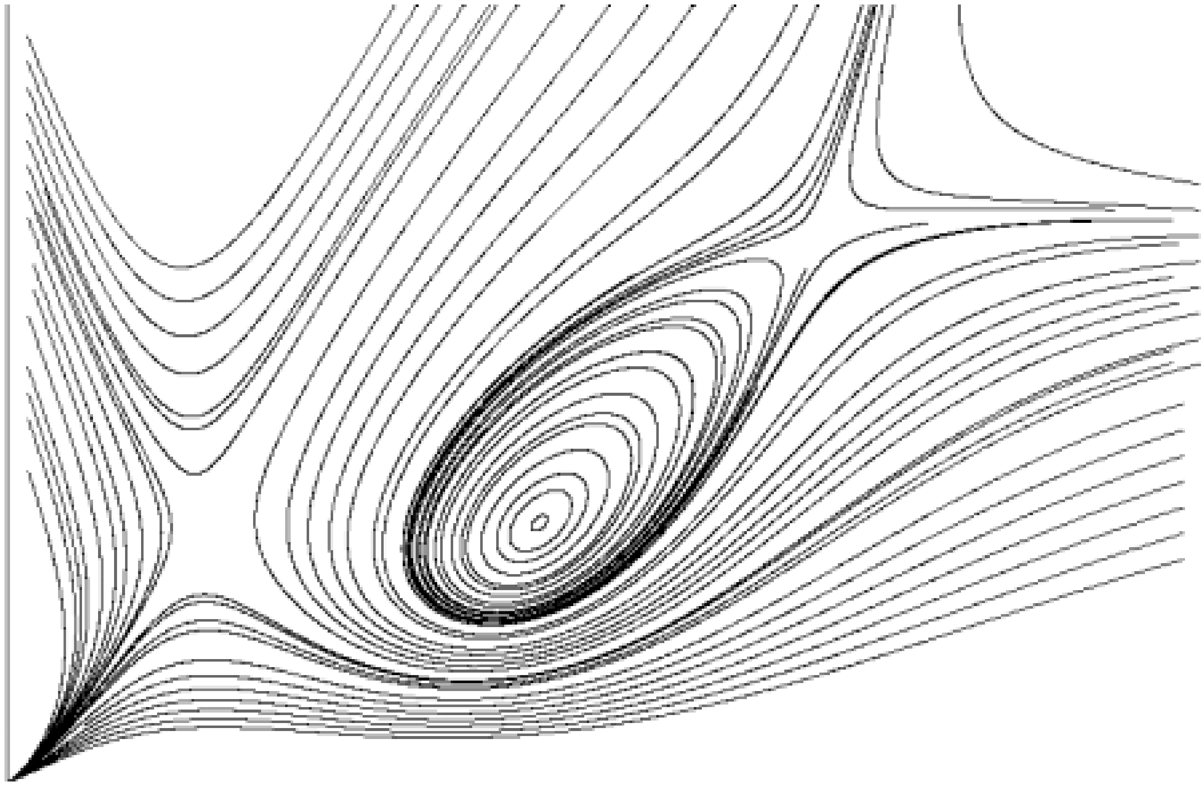


Figure 3c: Saddle point properties under open loop information structures

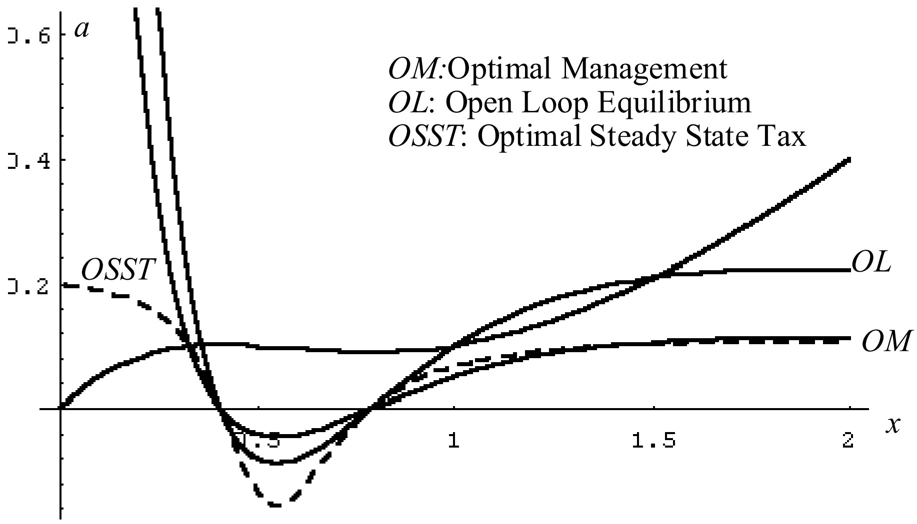


Figure 4a: Steady states under optimal management, open loop Nash equilibrium and optimal steady state tax, n=2

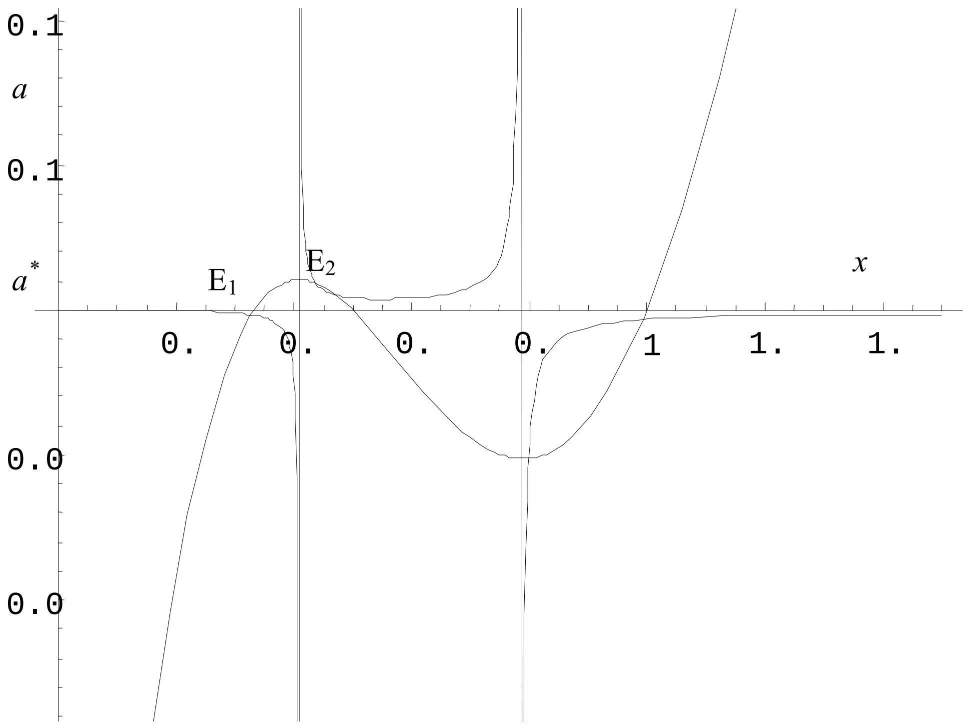


Figure 4c: Multiple steady states under OSST (n=500)

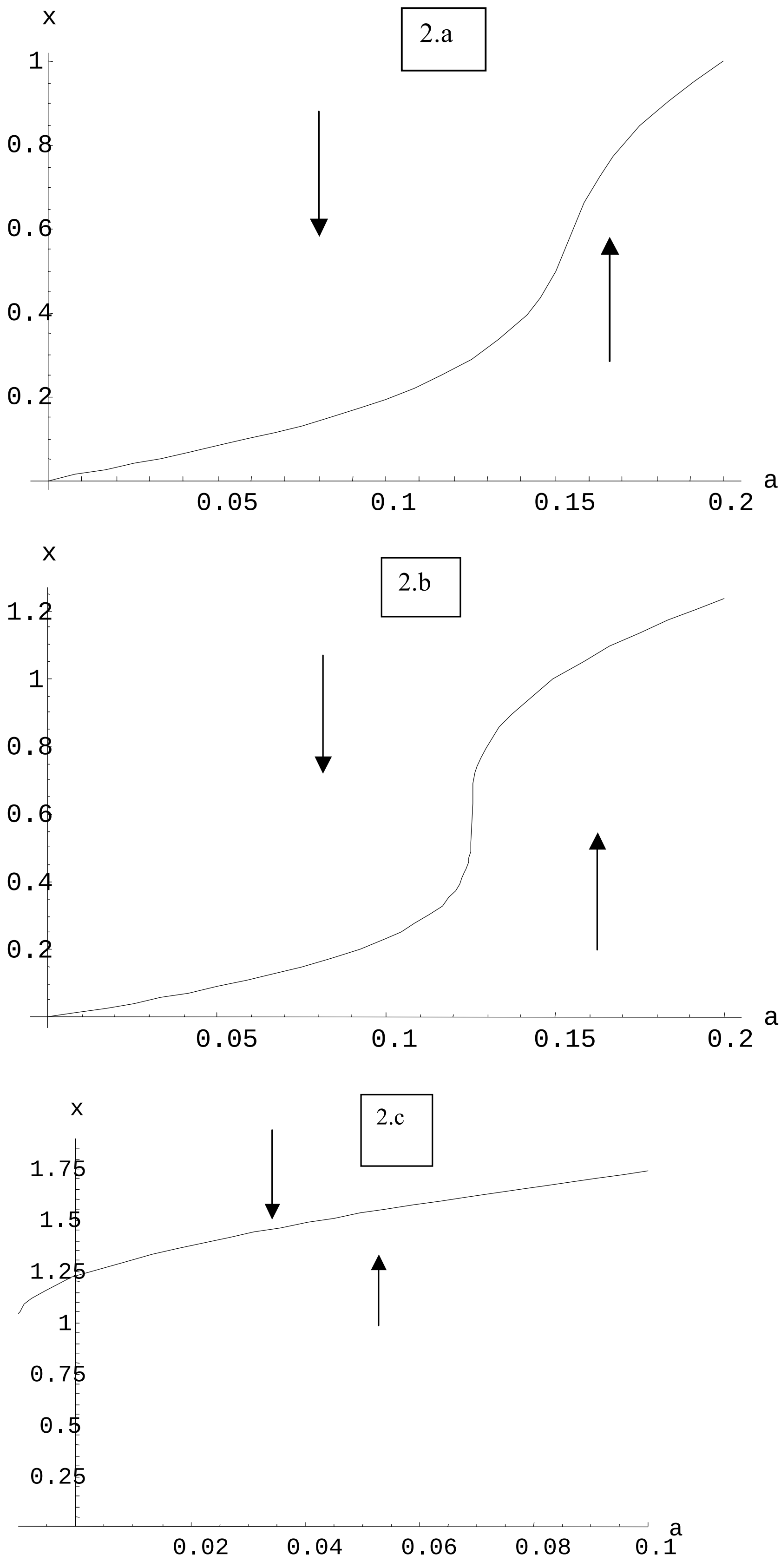


Figure 2: Hysteresis in non-linear ecosystems

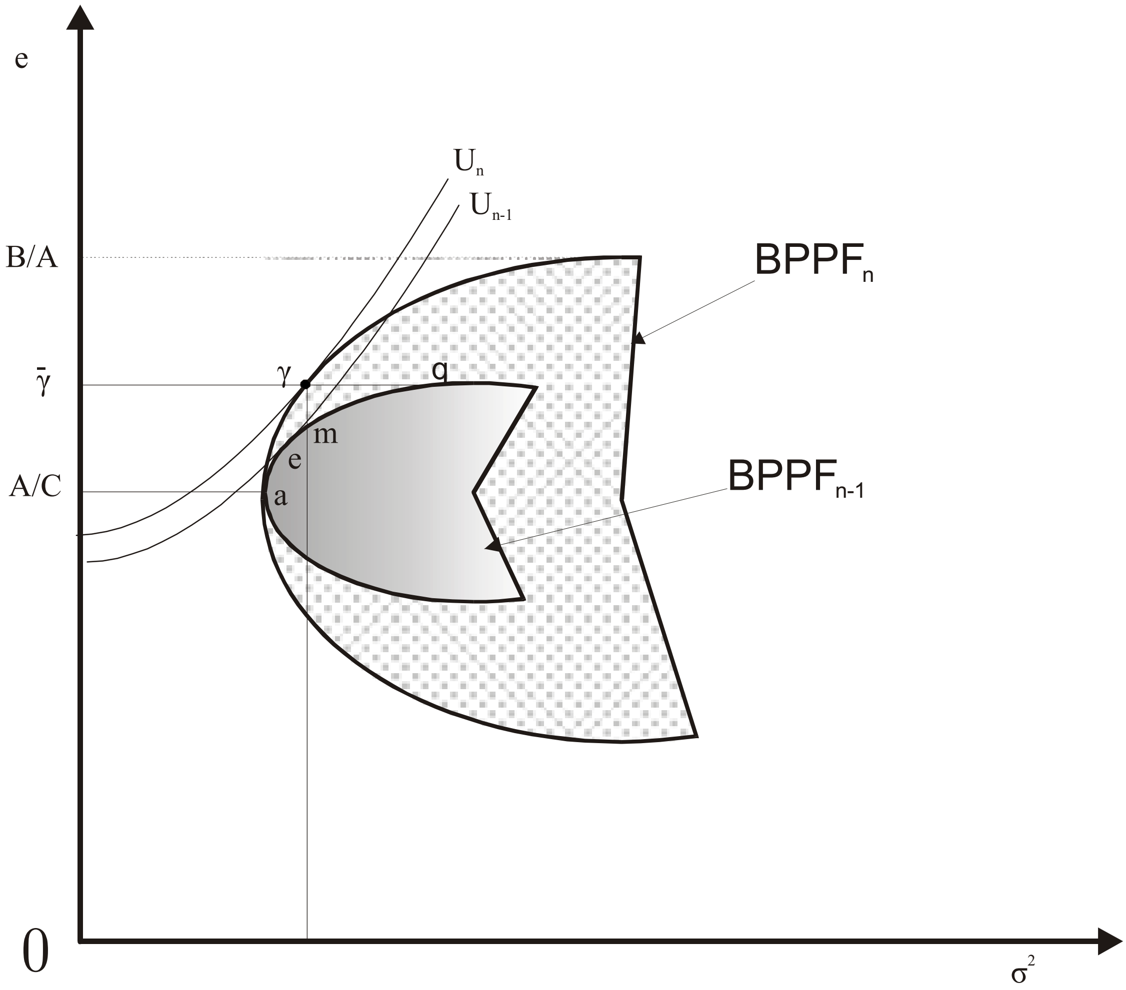


Figure 5: Biodiversity Valuation using CAPM methods

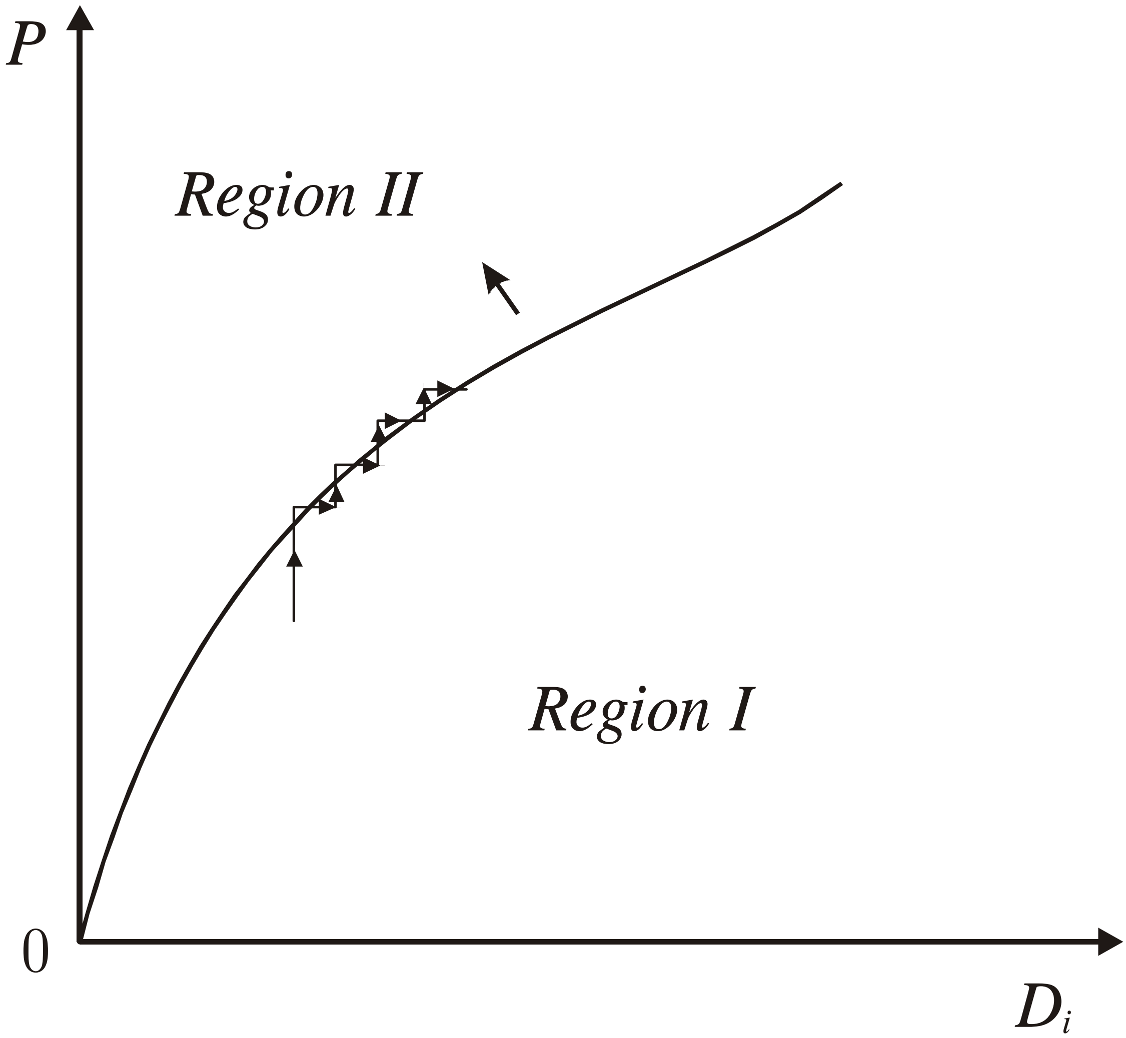
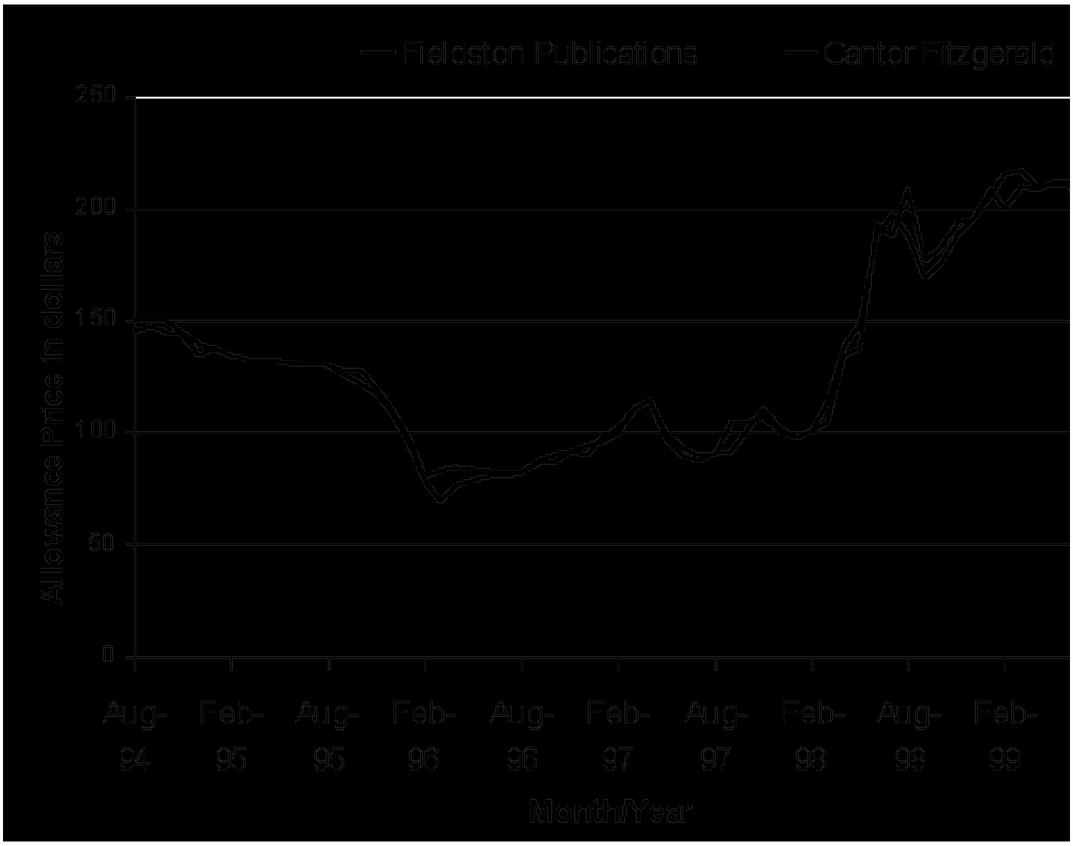


Figure 6: Barrier control policy

Monthly Average Price of Sulfur Dioxide Allowances
Under the Acid Rain Program



Though the EPA does not officially track allowance prices, the monthly average price of a current vintage year allowance, as reported by a brokerage firm and Fieldston Publications' market survey, is recorded here for informational purposes.

Graph 1: Monthly prices of SO_2 allowances (Source: EPA)