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Chapter 5

Goals, principles and constraints in environmental policies

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1. Introduction

This chapter explains how economists approach environmental policy questions and why their advise has not been translated into many applications. For many years there seemed to be a wide gap between what economists recommended and what policymakers were willing to accept and implement. While the former emphasized the need to arrive at decisions by carefully balancing costs and benefits, the latter paid more attention to social and political consequences. It is as late as in the 1990s that environmental economic analysts have started to address typical policymakers' concerns in a more systematic way resulting in a better understanding of economic considerations on the policy side.

The chapter consists of eight sections. Section two starts with definitions of three key economic concepts -- effectiveness, equity, and efficiency -- the last of which has been the traditional focus of economic analyses. It also defines the sustainability concept and relates it to the previous concepts. In the third section the notion of an economically optimal environmental policy is introduced. In the fourth section attention is turned to uncertainty and various social constraints that policymakers have to take into consideration. Several practical principles, including the well-known Polluter Pays Principle, which serve as useful rules of thumb are discussed next. Some additional questions on what and how environmental policies actually protect are raised in the sixth section. Finally the seventh section identifies typical policy failures encountered in real-world situations. The chapter ends with a brief assessment of the role played by economists in environmental policy debates.

2. Conflicting approaches to environmental policy

Like any other policy, an environmental policy has to compromise between different demands and expectations, many of them often conflicting with one another. Conservationists would like to protect natural habitats from economic development. Industrialists demand that protection measures do not hamper growth, and do not impose excessive burdens on firms. Social critics are concerned with fairness of the distribution of environmental protection costs among various groups and strata. In this regard three notions are crucial: effectiveness, efficiency and equity.

A policy is called effective, if it solves the problem it was supposed to. Thus the notion of effectiveness is closest to what would be the most likely focus of environmental activists and environmentally concerned citizens. Effective policies are those which clean the air, restore the lakes, and save species from extinction. The question of effectiveness does not refer to the costs such policies may imply, nor does it take into account any other social problems which may arise as a result of their implementation.

In contrast economists are concerned with the idea of efficiency attempting to take into account both costs and effects of a given policy or action. This implies that effects are made commensurate with costs by evaluating the former in the same terms as the latter. Monetary evaluation is the most obvious method to employ, but other -- less typical -- measurement units are possible too, eg energy units. A policy is said to be efficient, if its costs are justified in terms of its effects or, to put it more precisely, if it maximizes the positive difference between benefits and costs. As in the case of effectiveness, the idea of efficiency leaves aside the question of fairness, ie who will pay the costs, and who will benefit from the effects. Unlike effectiveness however efficiency addresses the question whether a policy is worthwhile. Thus an efficient air quality policy pushes emission abatement requirements up as long as incremental benefits resulting from the cleaner air exceed the costs of the cheapest alternative to meet such requirements.

Efficiency is a difficult concept to apply, as environmental benefits are often difficult to evaluate in economic terms (see Johansson, chapter 2 and Shechter, chapter 3 in this volume). This is why a somewhat less stringent concept of cost-effectiveness has been in common use. A cost-effective policy achieves any given effect at the least possible cost. Thus, if the objective is to cleanup a lake, of all effective policies the cost-effective one will be the one which restores the lake to life at minimum cost. It should be stressed here that neither effectiveness nor cost-effectiveness per se provide a criterion to judge whether a policy is worthwhile to pursue, ie, in the example above, whether the lake should be restored or the economy's scarce resources spent on something else. Efficiency provides a theoretical criterion, but of course there are additional aspects that environmental policy has to take into account.

Fairness has been another important issue raised in connection with environmental policies ever since such policies were formulated and implemented. There is no universally accepted definition of fairness, and economists prefer to talk about equity whenever they discuss the distribution of costs and benefits among parties concerned. Making a policy equitable means to balance costs and benefits across all parties concerned by appropriately distributing benefits and/or letting beneficiaries pay an adequate share in costs[ii]. For instance, a policy aimed at preservation of biodiversity will be judged inequitable, if the costs affect the local population in areas adjacent to protected habitats, eg by constraining development opportunities, without offering them a fair share in benefits from conservation.

As shown in the brief overview above, environmental policies can be viewed from several different perspectives. Traditional environmental activists would perhaps assess policies from the point of view of their effectiveness. More sophisticated activists would ask whether they are cost-effective, as the same effect can be achieved at various costs.

Economists, in turn, tend to raise efficiency questions. Although difficult to answer these questions are extremely important, because of two reasons. First, it is not sufficient to design a set of cost-effective policies to address environmental problems such as specific levels of acid-rain abatement, eutrophication control, solid waste disposal and so on. Even though each of these problems can be individually solved in the cheapest way, there is no reason to believe that the collective outcome will be what people would prefer to have, if you consider the costs to be borne. It might turn out, for instance, that the eutrophication control has become too strict *vis a vis* the waste disposal measures. It might have been better to relax the eutrophication controls a bit, and switch the resources saved in this way to substantially improving the waste disposal situation. Second, the resources spent on the environment altogether as a result of a series of cost-effective sectoral policies might turn out to be too few or too many in comparison with what was spent on meeting other needs.

In the 1980s long debates over alternative approaches to environmental policies led to the emergence of the sustainable development concept. Although there is no universal understanding of sustainability[iii], most definitions embrace the idea as expressed in the Brundtland report (WCED, 1987), ie to meet the needs of the present without compromising the ability of future generations to meet their own needs. Sustainability thus implies not only short-term equity (meeting the needs of the present), but also a much deeper concept of intergenerational equity[iv]. A <u>sustainable</u> policy has to address environmental problems in a way maintaining both the physical and social bases for further development. The idea of sustainability goes far beyond the scope of conventional environmental economics. It is difficult to operationalize, because much uncertainty exists whether the present use of natural and man-made resources is compatible with their future use. Regarded by many as cloudy, the idea of sustainability nevertheless serves as a guide for policymakers to balance their quest for economic efficiency with equity considerations, and to adopt a long-term perspective. It also helps to assess what is certainly <u>not</u> sustainable. In particular, with respect to the environment and natural use of resources, the sustainability concept directs the attention to the physical volume of inputs and outputs flowing through the world economy. This flow, conveniently called <u>throughput</u>, has been largely ignored by economic analyses, which focus on the monetary value of the selected components rather than on the physical scale[v]. However, because many important components are not captured by commercial transactions, monetary analyses often fail to recognize the important information derived from studies of the global throughput which, for obvious reasons, cannot grow indefinitely.

Despite theoretical and practical problems, there have been attempts to operationalize sustainable development by proposing rules on how to use the environment and natural resources in a sustainable way. The following three principles were suggested by Daly (1990, p.41).

1) With respect to the physical volume of inputs into the economy and its outputs: by consciously limiting the overall scale of resource use, shift technological progress from the current pattern of maximizing throughput to maximizing efficiency understood as the ratio of economic effects achievable from a given throughput.
2) With respect to renewable resources: by exploiting these on a profit maximizing sustained yield basis[vi] prevent them from driving to extinction. More specifically this means that:
   a)with respect to resources serving as inputs such as plants and animals, harvesting rates should not exceed regeneration rates;
   b)with respect to resources serving as 'sinks' such as the atmosphere of Earth, waste emissions should not exceed the renewable assimilative capacity.
3) With respect to exhaustible resources: maintain the total stock of natural capital by depleting nonrenewable natural components (such as mineral deposits) at a rate corresponding to the creation of renewable substitutes.

Principle (3) reflects what has been referred to as 'strong' sustainability, ie maintaining the total stock of natural capital rather than compensating for its depletion by investing in man-made capital. In contrast, 'weak' sustainability allows for substitution between the two types of capital, as long as the so-called Hartwick (1977) rule is satisfied: the revenues from depletion of exhaustible resources are invested rather than consumed. Here the natural capital can decline if man-made capital is increased by the same amount. Economists debate whether the two types of capital are substitutes or complements. While to some extent both types can certainly substitute for each other, it is obvious that the former provides necessary inputs to virtually all production processes. This suggests that 'strong' rather than 'weak' sustainability provides a more relevant reference for policymaking.

Various sustainability principles such as those mentioned above are very general, and do not give many practical clues on how to design environmental policies. Increasingly, however, more specific recommendations are being formulated, and their language becomes closer to that used by politicians and businessmen[vii]. What is even more important, the principles help to realize that environmental issues need to be seen in a much broader perspective than before. They are not just a matter of 'protection', but a matter of a long-term economic strategy.

3. An economically optimal environmental policy

If economists were to advise on designing environmental policies, the most likely suggestion would be to achieve efficiency by maximizing aggregate net benefits, ie the difference between total benefits and total costs (Baumol & Oates, 1988). In principle this approach needs not contradict equity considerations. For if a policy is efficient there is always a possibility to distribute its net benefits in such a way as to make everybody better off than in a non-efficient scenario. Moreover, by endowing property rights to individuals, it is theoretically possible to transfer costs and benefits across society according to almost any pattern. This makes a strong case for separating efficiency from equity concerns, and concentrate on efficiency.

However, studying real-life policy cases proves that the distribution of costs and benefits is most often ignored, and a situation of property rights from the pre-policy period is implicitly assumed. Not only may this imply that the benefits do not match the costs borne by various individuals, but also that some individuals are even made worse off. This is a typical outcome of many policies aimed at increased efficiency, especially in countries without adequate social security services. Thus the rationale of efficiency, ie the possibility of enjoying the maximized sum of net benefits, turns out to be a privilege distributed in a not necessarily fair way. Nevertheless many economists have viewed efficiency as an ideal reference point for designing sound environmental policies.

It is a tradition in economics to consider many important variables, including costs and benefits, as functions of a single variable under the control of a decision maker (see Hanley, chapter 4 in this volume). In the case of environmental policy the level of pollution or the level of exploitation of a natural resource, can serve as examples of such variables. It is then assumed that this variable can be flexibly determined by the policymaker so as to meet any criterion she may wish to choose. If the objective is to maximize the sum of net benefits, then one can apply the following argument. Choose any initial level of the control variable, and ask whether increasing its value by a small amount would imply more benefits or more costs. These incremental benefits are called Marginal Benefits (MB). Likewise, incremental costs are called Marginal Costs (MC). As long as MB>MC, it is worthwhile to increase the level of the control variable. If one finds that MB<MC, then it pays to decrease the level, since what one loses in terms of foregone benefits (MB) one more than compensates in terms of avoided costs (MC). Thus, unless one hits a boundary of the control variable's domain earlier, the only point where the net benefits are maximized is when MB=MC. The important corollary is that an efficient policy should aim at equating marginal costs with marginal benefits of environmental protection.

The MB=MC criterion has largely remained a theoretical reference point for various policy instruments (see Löfgren, chapter 1 and Barde, chapter 6 in this volume).

Time dimension can be added to this analysis by introducing a rate of discount. To many people, discounting future values is unfair and arbitrary. Yet without discounting, intertemporal choices would be difficult to make. The rationale for discounting results from a so-called time preference. Let us assume that there is no inflation (which does not change the idea but makes the calculations less complex). Most people are not indifferent to getting either \$1000 today or \$1000 a year from now; they would prefer to have it sooner rather than later. But how about having \$1000 now or \$1500 a year from now? Most of us would probably prefer the latter option. Perhaps there is some amount of money $\$1000(1+\delta)$ between \$1000 and \$1500 that makes us indifferent to having either \$1000 now or $\$1000(1+\delta)$ a year from now. The number δ which renders the two options equivalent is called the rate of time preference. It is a fundamental component of any discount rate used in order to compare costs and benefits accruing at different points in time. If $\delta=0.025$ (2.5%) then we would consider \$1025 a year from now as equivalent to \$1000 now and \$1000 a year from now is equivalent to $\$1000/(1+\delta)$, that is, approximately \$976 today.

It is easy to extend this concept to time intervals of any length. The present value of \$1000 two years from now is $\$1000/(1+\delta)^2$ and so on. If a project requires costs of 1000, 100 and 200 now, a year from now and three years from now, respectively, then its discounted sum of costs is $\$1000 + \$100/(1+\delta) + \$200/(1+\delta)^3$. If it provides the investor with benefits of \$300, \$400, \$400, and \$300 after the first, second, third, and fourth year, respectively, its discounted sum of benefits is $\$300/(1+\delta) + \$400/(1+\delta)^2 + \$400/(1+\delta)^3 + \$300/(1+\delta)^4$. The net present value, NPV -- a key concept used in cost-benefit analysis -- is the difference between the discounted sums of benefits and costs. As before, the MB=MC criterion (with discounting used to account for the time dimension of benefits and costs) helps to identify maximum NPV.

Substituting 2.5% for δ in the example above would yield NPV equal \$33.35. Mere subtraction of (undiscounted) costs from (undiscounted) benefits would give the difference of \$100. This can be interpreted as NPV with zero discount rate. Thus discounting with positive rates decreases the value of projects whose costs come earlier than benefits. The same example recalculated with 5% discount rate will render a negative NPV.

Most projects require costs to be borne before benefits can be enjoyed. Environmental projects are often characterized by high costs and by benefits extending over a long period of time. Or the time lapse between launching a project and reaping its fruits can be long.

The conclusion some people draw from these facts is that discounting is anti-environmental and anti-sustainable. Indeed, high discount rates may imply negative NPV for projects with benefits that are modest but sustainable over a long period of time. Applying a zero discount rate, however, is not a good solution. Firstly, it is incorrect since people do reveal time preference. Secondly, it is also counter-productive from the environmental point of view; zero or low discount rates favour excessive investment, leading to an increased throughput, with all its risk of resource exhaustion and environmental degradation and the waste of capital. Most economists agree on the need for applying realistic discount rates. At the same time, they indicate that it is more appropriate to address sustainability concerns directly rather than by ignoring or underestimating the time preference of people.[viii]

In some analyses economists apply an abstract measure of benefits net of costs, called <u>utility</u>. It can then assumed that welfare is improved if aggregate utility is increased. However, in order to make intertemporal comparisons, future utilities need to be discounted. Thus a welfare maximization criterion reads:

$$\Sigma_{t=0}^{\infty}u_t/(1+\delta)^t, \tag{1}$$

where u_t is the utility in the year t and, as before, δ is a discount rate.

Formula (1) attaches declining weighs to the welfare of future generations. Thus a development scenario transferring wealth from the future to the present would yield a higher NPV than one keeping u_t constant (or rising). Hence maximizing a flow of discounted welfare may contradict the definition of sustainability introduced in the previous section. On the other hand, this definition is sometimes seen as too rigid to reflect all scenarios which are discussed in actual policy debates and meet some long-term fairness criteria at the same time. It was proved that under reasonable assumptions (Chichilnisky 1996) any notion of long term 'optimality' is equivalent to maximizing the following formula, often referred to as the <u>Chichilnisky criterion</u>:

$$\alpha\Sigma_{t=0}^{\infty}u_t/(1+\delta)^t+(1-\alpha)\lim_{t\to\infty}u_t, \text{ for some } \alpha\in[0,1] \tag{2}$$

The second component in this sum represents the undiscounted welfare of a single generation in a distant future. Of course, for $\alpha=1$ the Chichilnisky criterion boils down to formula (1). On the contrary, for $\alpha=0$, the intergenerational distribution of utility does not count, as long as welfare is maximized for generations that are far ahead. Intermediate αs correspond to combinations of NPV and the future welfare condition.

In order to derive further implications of formula (2), one needs to make additional assumptions. First, let us define u as a function of consumption c and the state of the environment s, $u_t = u(c_t,s_t)$. Second, let the relationship between c and s be given by the following equation:

$$s_{t+1} = s_t+r(s_t)-c_t, \text{ and } s_0 \text{ is given,} \tag{3}$$

where r is the natural regeneration rate. If $r\equiv0$ then s should be interpreted as an exhaustible resource whose stock can only decline over time. If $r>0$ then s corresponds to the stock of a renewable resource which can be kept constant if $c_t=r(s_t)$ for all $t\geq0$ (ie when the consumption equals the regeneration rate).

Two rather unexpected theorems can be proved[ix]. First, for any $\alpha\in(0,1)$, and any constant $\delta>0$ the problem:

$$\begin{aligned}&\text{Max}_c\ \alpha\Sigma_{t=0}^{\infty}u(c_t,s_t)/(1+\delta)^t+(1-\alpha)\lim_{t\to\infty}u(c_t,s_t),\\ &\text{subject to: } s_{t+1} = s_t+r(s_t)-c_t, \text{ with } s_0 \text{ given}\end{aligned} \tag{4}$$

does not have a solution. Second, if the discount rate goes to zero with time, ie $\lim_{t\to\infty}\delta_t=0$, then the problem above has a solution; the solution is exactly the same as for the NPV maximization (with a declining rate of discount):

$$\begin{aligned}&\text{Max}_c\ \Sigma_{t=0}^{\infty}u(c_t,s_t)/(1+\delta_t)^t,\\ &\text{subject to: } s_{t+1} = s_t+r(s_t)-c_t, \text{ with } s_0 \text{ given.}\end{aligned} \tag{5}$$

The intuition behind the first theorem is that increasing consumption temporarily -- resulting in its temporary decline later on -- may increase the first operand of the addition in formula (4) without affecting the asymptotic behaviour, ie the second operand in (4). What it says in plain language is that discounting with a constant positive rate cannot be reconciled with keeping future generations at least as well off as the present. Then the second theorem says that the reconciliation can be reached by applying a variable discount rate which tends to zero over time. But does such a discount rate make economic sense at all?

It was observed earlier that people do apply positive discount rates when making decisions about the future. However, there is empirical evidence that the discount rates applied to future projects tend to decline with the futurity of these projects. In addition, there is also some theoretical justification for this phenomenon (Beltratti et al. 1998, pp.63-64). Thus testing for sustainability of a policy may apply standard cost-benefit analysis tools provided that implications of a very long run perspective -- such as declining discount rates -- are taken care of.

4. Information and uncertainty

One reason why policymakers have not so far followed economists' prescriptions is that any estimates of marginal costs and marginal benefits are affected by a wide margin of uncertainty. Although for different reasons, neither benefits nor costs are usually known with accuracy so that often the MB=MC rule cannot be adopted as a practical guide even in a short run. In a long run additional problems result from ambiguities affecting discount rates.

Environmental standards and other regulations are most often justified in non-economic terms. This should not be a surprise, as it is impossible to attach a price tag to everything. How could one put a price on human life and health constantly threatened by various man-made environmental dangers? How could one convincingly price the Acropolis of Athens now dissolved by acid rain? What is the value of saving a species from extinction? Even though economists are prepared to give answers to all of these questions, they must admit that both their methodologies and the accuracy of their measurements are disputable (see Johansson, chapter 2 and Shechter, chapter 3 in this volume). In such circumstances it is more practical to check policy measures against non-economic criteria too.

In addition, there are instances where uncertainty does not come from economics, but rather from natural sciences. Chains of cause-effect relationships leading from anthropogenic impacts to environmental changes, and finally to human health and welfare consequences are often not known in detail. Thus, reliance on too sophisticated models may more easily lead to blunders than basing the policy on common sense and on some simple rules of thumb. Engineers in designing bridges apply margins for uncertainty; perhaps environmental planners should do the same in certain cases. This is called 'safe minimum standards' or 'prudent estimates of critical loads'.

Quite different sorts of uncertainty affect the cost side. There are few, if any, pricing controversies. The main problem arises in connection with assessing what costs should be viewed as expenditures inevitable in meeting environmental requirements. It is easy to point at large amounts of money allegedly needed to comply with a regulation; industrial lobbies raise this argument continuously. Of course the argument is based on the extra cost of an 'end-of-pipe' treatment on top of an existing technological process. Indeed, control technologies may be costly, but there are often cheaper alternatives available.

First of all, process-integrated refinements aimed at better utilization of inputs lead to fewer waste products, and improve the environmental performance of a firm almost for free. It turns out that such changes may even lower production costs in some cases, and environmental requirements thus help to stimulate purely commercial innovation (see Corbett & van Wassenhove, chapter 10 in this volume). For instance the global program for phasing out CFCs revealed that their substitutes in certain applications are not only less harmful, but also cheaper (see Gabel & Sinclair-Desgagne, chapter 9 in this volume). Sometimes it is possible to decrease emissions by switching to another input mix or by changing product design. Lastly, there is always an option to scale down production, relocate production facilities, or simply shut down a plant. The last option may prove to be a viable solution if the plant was barely making a profit even in the absence of strict environmental regulations. All in all, before a costly 'end-of-pipe' equipment is installed, a number of more intelligent alternatives ought to be screened. Further, public interest in a region can be served best, if the decision on who should abate how much is arrived at after determining who offers the cheapest options.

No firm will share information on the true costs of doing business. Such a disclosure would affect its strategic position *vis a vis* its competitors. Information on abatement costs would additionally affect its position *vis a vis* a regulatory agency by constraining its bargaining opportunities over environmental requirements. It is quite natural that industries -- in their own interest -- overstate future compliance costs. No government nor any other external body can unravel all the options that are open to a firm. After all it was ultimately the lack of entrepreneurship which caused the centrally planned systems in Europe, attempting to substitute administrative coordination for free enterprise, to collapse. The lesson learnt from their failure is relevant for environmental policy too.

The pursuit of self-interest can be used as a guide to lowering environmental protection costs, and the market is the best mechanism to reveal what expenditures are really inevitable to meet environmental requirements. Markets can reveal what is the cost society really has to bear to meet these requirements. Then, in turn, economic instruments can 'harness' market forces to work for the environment's sake. Economic instruments perform a number of useful functions. But their most important task is to minimize overall costs of environmental protection through a cost-effective differentiation of control requirements. To be precise, those agents with the lowest abatement costs should be given the most stringent requirements. As argued above, for obvious reasons no agent, whether a firm or a consumer, would provide the regulatory agency with the information needed to take such decisions. Arriving at least-cost solutions thus requires that market forces are set to work, and take advantage of the cost-saving potential of economic instruments.

Environmentalists know, however, that markets have been a mixed blessing. Indeed, a lot of harm is done within the framework of thriving market forces both in the developed and the less developed world. It is therefore necessary to assess as objectively as possible what can be expected from markets, what information they can successfully provide, and where they will not be helpful at all. The distinction between scale and allocation decisions has proved to be a useful methodological starting point.

To regulate the environment means to decide first to what extent a given resource, say the carrying capacity of the atmosphere or an aquifer, could be used; this is called the 'environmental utilization space', Opschoor (1992) or scale Daly (1992); and, second, what portion of the resource should be allocated to any of its potential users. The first decision addresses the <u>scale</u> aspect, the second the <u>allocation</u> aspect. To quote a specific example: millions of polluters contribute to the eutrophication of the Baltic Sea. If the sea is to be restored, the influx of nutrients (phosphorus and nitrogen) has to be decreased. It has been estimated that bringing the total influx down to the level observed in the 1950s would do the job (Wulff & Niemi, 1992). This can be done in a number of ways. One way would be to require that every polluter decreases the nutrient discharges by a uniform rate to meet the overall objective. But certainly there are many other ways to allocate requirements between various regions, economic sectors and individual sources (Gren et al. 1997). The first (hypothetical) decision to reduce the overall influx of nutrients is an example of a scale decision. It provides a framework for another decision, that of allocation ie, on how to distribute tasks among all the parties concerned.

These two aspects can be addressed separately. But they can also be treated jointly by starting with the allocation of tasks thus arriving at the overall scale as a result of individual contributions. Also market forces can be utilized to determine either of the aspects. However the role markets can play in either case may vary.

A competitive market is a superior mechanism for calculating equilibrium prices by revealing people's preferences and unravelling what entails the least costs of providing goods or services. In this capacity it has never been surpassed by any other institution. However markets cannot do impossible things. For instance, they cannot process information which does not exist. In particular they cannot estimate the value of an environmental resource, if the resource is not freely exchanged and/or if its contribution to human welfare, both direct and indirect through its role in a wider ecological-economic system, is not fully known. Likewise they cannot convincingly evaluate benefits which are future benefits, especially those to people that do not exist yet. Hence, while it is quite legitimate and rewarding to employ the market as an allocation mechanism, it would be unreasonable to rely on it in determining scale decisions.

This separation principle is well illustrated by marketable permits (see Barde, chapter 6 in this volume). The volume of all permits issued determines the scale aspect of a policy. But the allocation is left to market forces either by auctioning the permits or by making permits transferable after distributing them in an administrative way. Marketable permits utilize market forces exactly in the domain where these can play an outstanding role, and keep them out from where they may sometimes do more harm than good. Note that the choice between marketable permits and other economic instruments such as charges or taxes, is a choice between assigning priority to economy or to environment. In an uncertain world, with taxes the financial outcome of environmental policy is less likely to cause surprise. Whereas with marketable permits the scale of protection is less likely to cause surprise (see Löfgren, chapter 1 and Barde, chapter 6 in this volume).

Dealing with uncertainty has led policy analysts and politicians to borrow some concepts from the theory of games. To play a game means to take the chance of losing while expecting to win. Game theorists have developed an elaborate conceptual framework to assess and prioritize strategies given their adequacy under alternative scenarios. As a rule, strategies which are promising from one point of view are inappropriate from another one. For instance, if one pursues a development policy aimed at short-term financial profits, then one may be surprised with its adverse health or environmental effects. And vice versa, an environmentally effective policy may turn out to be an unexpected burden, especially if judged from a short-term perspective. Are there strategies which do not sacrifice one goal for another in an uncertain world? In many applications there are and they are called win-win or no-regret strategies. An example of such a strategy would be to invest in projects which result in economic development and environmental improvements simultaneously.

The no-regret approach to environmental policy is another recent contribution to empirically assessing alternative policy measures. It calls for adopting not necessarily those ones which are superior with respect to the theoretical criterion of efficiency, but those which are likely to prove robust when tested against a mix of other criteria as well. The World Bank for instance has identified a number of no-regret policies which serve both economic development and a better environment. They include poverty reduction through a redistribution of wealth, elimination of subsidies for the use of fossil fuels and water, improved sanitation, endowing farmers with property rights on the land they farm and so on (World Bank, 1992). This does not mean that there is no trade-off between environmental quality and the level of material production, or between short-term increase in production and the ability to sustain this level over a longer period of time. It simply means that it is sometimes possible to eliminate false dichotomies by screening options carefully and selecting measures which address several concerns at the same time.

5. Popular principles in environmental policy

As pure theoretical criteria, such as efficiency or cost-effectiveness, are difficult to apply in real-world situations, environmental policies have often referred to several simple practical principles. These include such well-known maxims as 'Polluter Pays', 'Polluters Pay', 'User Pays', 'Victim Pays', 'The Precautionary Principle', and 'The Subsidiarity Principle'. Although frequently quoted, these principles nevertheless are by far not obvious to everyone, and there are controversies about their exact meaning.

Polluter Pays Principle (PPP) is probably the best known guideline for environmental policy (OECD, 1992). There are, however, two definitions in use. PPP in the broadest sense means that the polluter is financially responsible for whatever harm its activities may cause, no matter whether they stay within the limits set by the law or not. Even though such a broad range of responsibilities have been theoretically possible under jurisdictions adopted in many countries and encouraged by the OECD since 1991, most policies try to enforce a narrower version. PPP in a strict sense means that the polluter is financially responsible for complying with whatever environmental requirements are set by relevant authorities.

As it turns out, even this more restricted form of PPP, officially endorsed by the OECD since 1972, has been difficult to apply in practice. An alternative wording of PPP would be 'No-Subsidy Principle'. But important exceptions to this principle are acknowledged and tolerated by the OECD. Subsidies for polluters to meet environmental requirements are approved if the following three conditions are met: 1) the subsidy does not introduce significant distortions in international trade and investment[x]; 2) without the subsidy, affected industries would suffer severe difficulties; and 3) the subsidy is limited to a well-defined transition period adapted to the specific socio-economic problems associated with the implementation of a country's environmental policy[xi] (OECD, 1992).

Some governments have developed a very peculiar understanding of the PPP (Opschoor & Vos, 1989, pp. 28-30). The Federal government of Germany, for instance, considers certain investment subsidies compatible with the PPP as long as polluters are responsible for the rest of abatement efforts, including, most importantly, operation costs. On the other hand, the Swedish government extends the PPP to justify introducing 'control charges': raising funds to more effectively regulate the polluters and monitor their compliance. Some governments admit that they violate the PPP for equity reasons if the polluter is a municipal body, as in the case of the United States Construction Grant Program to subsidize sewage treatment in urban agglomerations.

As the PPP has been frequently misinterpreted and misunderstood, it is important to define what this concept actually encompasses and what it does not[xii]. First of all it was established 'to avoid distortions in international trade and investment' (OECD, 1992), which may occur when polluters are subsidized. Seen from such a perspective, the PPP offers a reasonable and appealing common cost allocation principle. The choice of this particular principle was motivated by economic considerations. These suggest that once prices of goods and services reflect their full social costs, including environmental ones, market forces start to work for a good purpose, and less intervention is necessary to achieve the desired environmental effects.

There are three features which are often associated with the PPP, even though they should not. First, PPP is not an equity-motivated principle. One may judge it equitable that the polluter pays but who eventually bears the burden of protective measures is a different question.[xiii] As a matter of fact most deliberate violations of the PPP, such as subsidizing the construction of municipal sewage treatment plants, have been motivated by equity concerns. Second, PPP is not a liability principle. It is not a matter of designating who is responsible for pollution: whether the supplier of an environment-unfriendly product (eg the producer of a pesticide) or the user (eg the farmer). It is merely a matter of determining at what level it is most appropriate to account for environmental costs and/or to step in with regulations. Third, PPP is not synonymous with environmental taxation. It can be implemented in many different ways, ranging from pollution charges to direct regulations, as long as polluters are forced to take protective measures (see also Barde, chapter 6 in this volume).

When strict financial responsibility of a polluter is difficult or impractical to enforce, governments sometimes apply the Polluters Pay Principle (note the plural form!) charging polluters the environmental protection costs at large. This principle assumes that polluters are charged more or less in proportion to the environmental stress they cause, in order to raise funds to support protection activities needed to comply with regulations. The French pollution charges recirculated, ie paid back to the polluters to subsidize their abatement activities, are a well-analyzed example of that sort. Also in the Netherlands there have been similar programs in operation, although the Dutch authorities prefer to point at the so-called causation principle as a justification for their charging polluters at large, and recirculating money for environmental control purposes. In Poland, the funds originating from pollution charges contribute 40% of the overall investment expenditures, and they have become an important component of the country's recovery program (Zylicz, 1998).

Both 'Polluter Pays' and 'Polluters Pay' should be distinguished from a simple 'User Pays Principle'. Here, environmental protection is achieved through the operation of a facility eg sewage treatment plant or landfill, serving a group of users. If the facility is financially self-sufficient, then it can be said that the User Pays Principle applies, ie the users pay for its operation. Even though the concept seems to be clear at first glance, ambiguity may arise when the costs of a facility are not or cannot be divided among the users proportionally, because of the heterogeneity of services required. This happens if, for instance, there is no obvious way to assign the overhead costs of the facility to what various users discharge. Cross-subsidies, ie subsidies of one category of users to another, may happen as a result. Sometimes the cross-subsidy is deliberate, eg charging industrial users for water supply more than municipal ones, and serves an equity purpose. In such cases, it would be more appropriate to talk about a 'Users Pay Principle', but this term is not used.

Like the 'Polluter Pays', the 'User Pays' is basically a 'No-Subsidy Principle'. But in many countries and in many instances they have been violated due to equity considerations. This is somewhat paradoxical, since these principles were often primarily warranted not so much on efficiency as on equity grounds. It has been known since the 1960s that in a two-party setting efficiency does not require the polluter to be charged for the harm done to the victim. What is important is that the property rights to the environment are assigned unambiguously and can be enforced, such that the polluter and victim are prepared to negotiate, and that the transaction costs of any potential protection agreements are negligible[xiv]. Thus sometimes the efficiency question can be disconnected from the allocation of protection costs: these can either be born by the polluting or by the polluted party. If for equity reasons (or for any other reason) the polluter is not expected to pay, then efficiency can be achieved by letting the 'pollutee' subsidize ('bribe') the other party not to pollute. This is referred to as the Victim Pays Principle. It has been widely debated,[xv] and occasionally applied in agreements to curb transboundary pollution (Zylicz, 1991).

Even though short-term efficiency may often be achieved irrespective of who shall bear the responsibility to pay protection costs, it is rather obvious that in the long run it is the polluter, or the user who should bear the burden. Otherwise they are in fact subsidized which will direct economic activities to the domain where they cannot be sustained, and will remove an incentive for environment-saving technological innovations. For these reasons, the 'Polluter Pays', and 'User Pays' should be seen as useful policy guidelines.

Apart from the cost-related principles discussed above, there are others inspired by non-financial considerations as well.

As for the 'no-regret' approach, the motivation for The Precautionary Principle can be derived from game theory concepts and corresponds to the so-called minimax strategy, ie a strategy to minimize the worst possible outcome. This principle is nothing more than a general recommendation to expect an unfavourable course of events, and to choose policy measures accordingly. However, it has several useful consequences. One is to establish 'safe minimum standards' (see section 3 above) when dealing with materials or technologies whose environmental and health effects have not been fully researched yet. Such standards, based on whatever fragmentary evidence is available, should prevent major surprises even if the worst is going to happen.

An example of another corollary derived from The Precautionary Principle is the use of financial guarantees for the worst possible outcome. These can be gradually lowered subject to further evidence reducing the likelihood of a worst-case scenario. This is the rationale behind 'environmental performance bonds' suggested mandatory for agents entering potentially hazardous activities. An environmental performance bond is an insurance, to be bought by an economic agent willing to enter an activity which may lead to serious adverse environmental effects, whose price depends on the scientific assessment of risks involved (Perrings, 1989). Insurance provides funds for remediation if necessary. But an even more interesting feature of this instrument is the prospect of linking the price of the bond to the on-going evaluation of the agent's performance including the ability to convince the authorities (or the general public) of the safety of its operations. Thus this can also contribute to the technological and scientific progress aimed at reducing uncertainty.

The Subsidiarity Principle is a postulate originating from political and social considerations. It states that policy measures should be determined by the lowest level of authority suited for a given problem. In other words, it can be seen as a 'Decentralization Principle'. In the European Union it has been applied to let the member states adapt their policies to the Union's directives in a creative way, and make them compatible with local preferences[xvi]. From a sociological point of view this principle is much more than a simple safety-valve against the omnipotence of centralized bureaucracies. It is a recognition of the fact that people do not only have different preferences but also aspirations to monitor the resources they are responsible for. 'The Subsidiarity Principle' can be linked to the deeper concept of regionalism ie, an 'approach to the study of society as based on the recognition of distinct differences in both cultural and natural attributes of different areas that, nevertheless, are interdependent' (Odum, 1983, p.512). The consequences of adopting a regional outlook are twofold. First, because of the uniqueness of their natural attributes, different regions may require different 'customized' measures, standards and policies[xvii]. Second, these policies should treat the region as a whole, and serve both environmental and human needs.

The regional approach can be viewed as yet another manifestation of the sustainable development philosophy. It would be difficult, and also not sustainable, to promote policies which do not take into account local aspirations. Such policies would be either poorly enforced or, if effectively enforced against the will of the local people, would be a force disrupting the stability of the political structure which implemented them. Non-sustainability would then result from impairing either the natural or social base of the region's development. On the other hand however, as observed by Odum above, regions are interdependent and the decisions taken by one region may affect any other region. If there is a possibility for a region to 'free-ride' on others, or to dispose of a problem through creating an 'externality' (see Löfgren, chapter 1 and Folmer & de Zeeuw, chapter 17 in this volume), it is in the interest of all regions concerned to exert a mutual coercion jointly agreed upon. 'The Subsidiarity Principle' always has to be judged against specific considerations that may require a supra-regional authority for the common good.

Similar qualifications apply to any of the principles discussed above. They were developed as useful rules of thumb to point at options which are adequate most of the time, or in nearly all typical circumstances. It was made clear in this chapter, however, that there is no underlying theory implying that these principles are generally valid rules of successful environmental policy-making.

6. What to protect and how?

Two important technical questions have to be answered before an environmental policy can be drafted. These are: 1) Who or what shall we protect? Should we protect the environment for ourselves, or rather should we protect ourselves against possible hazards scattered throughout the environment? In the latter case, do we want to protect an average individual or the most susceptible subpopulation? 2) How far should we intervene in spontaneous economic development processes? In particular, should we regulate industries by imposing requirements derived from the best-known abatement technologies, or by imposing requirements derived from an assessment of the environmental 'carrying capacity' or utilization space? The most likely answer of economists to both questions would be to estimate costs and benefits of each alternative and to select the most efficient one. As indicated in section 3 above, this is not the approach that has been followed in practice. Environmental policymakers cannot escape making hard social choices and explicit political decisions by simply referring to the outcome of a cost-benefit analysis.

There are two broad orientations with respect to the first question mentioned above: the environmental and the health approach. Within the former, priority is given to environmental considerations. These may turn out to be more restrictive than the health aspects. For instance, coniferous forests are more vulnerable to acid rain, than humans are.[xviii] Thus an environment-oriented ambient standard for sulphur dioxide will be stricter than a health-oriented one. Also, within the realm of natural ecosystems there are some that are more vulnerable than others. Looking at the most fragile one would probably require the total elimination of a pollutant or the reduction of it to the natural background concentrations, if it is a naturally occurring substance, like sulphur dioxide.

It is very unlikely, though, that environmental policy would adopt the general rule to protect the most fragile ecosystem. Likewise, for practical reasons, it is unlikely that within the framework of the health-orientation, it will follow the rule to protect the most vulnerable subpopulation. Rather one can observe that a more pragmatic approach is adopted of setting ambient requirements so as to protect a typical ecosystem considered to be the most important one, and/or an average individual. With respect to the most vulnerable ones (both ecosystems and humans) the best a policymaker can do is to protect them from the environment. This strategy is practiced by establishing rescue programs for threatened species, creating gene banks and so on. Advising asthmatics to stay indoors during smog alerts, or excluding certain sites from being visited by pregnant women and children, as well as by other high risk groups, serve a similar purpose. This is by far not a recommended solution, but in some instances, at least in the short run, there are no other politically viable methods to proceed.

The second important question is whether to do what is technologically possible, or what is environmentally desirable. It would be optimal to have an easy technological solution to every environmental problem. Unfortunately this is not the case, and environmental or health requirements imply constraints on economic development. What is environmentally desirable may not be technologically possible.

The problem can be best explained in the context of the distribution of emission permits among polluters affecting air quality in a given area. Each of these polluters has a certain production plan leading to certain pollution loads unless abatement measures are enforced. Which measures should be imposed?

One approach is to require everyone to apply the very best abatement technology. This is how such mysterious acronyms as BACT, RACT or BATNEEC were invented and used in some countries. They stand for Best Available Control Technology, Reasonably Available Control Technology and Best Applicable Technology Not Entailing Excessive Costs respectively. Thus some environmental policies have relied on the requirement that every source is equipped with a state-of-the-art technology. This is the <u>source- or technology-based approach</u>. Standards based on this are called <u>emission or source-oriented standards</u>. This philosophy has a few advantages, the most important of which being simplicity and appeal to the common sense: everybody does what can, and thus ought to, be done.

It has several drawbacks. First, requiring that everybody does what is technologically possible, even in the more liberal BATNEEC version, may not be cost-effective, since different polluters will probably end up with different marginal costs of abatement. Second, if the area of concern is filled with a large number of polluters, the overall impact on the air quality can be intolerable, even if every plant was equipped with a BACT device. The technology-based approach may thus violate not only the cost-effectiveness, but also the effectiveness criterion. The same argument certainly applies to consumer behaviour. If for instance the number of cars becomes too large, then even if each of them was equipped with a frontier technology, the total emissions could be excessive.

Despite such objections, in 1996, the European Union adopted the Integrated Pollution Prevention and Control (IPPC) Directive mandating emission sources to apply Best Available Technique (BAT). Even though the Directive does not prescribe a specific technology, it requires emission to be regulated -- source by source -- up to the level that corresponds to an existing technical option. There is no mechanism to achieve cost-effectiveness under the IPPC Directive.

An alternative approach is to ignore the existing polluters, and to define the desired air quality target level. This represents the <u>ambient quality approach</u>, standards based on which are called <u>ambient standards</u>. Thus an environmental policy may start with an ambient standard which is translated into total emissions allowed in the region in order to meet this standard. It may turn out that production plans of industries together with consumers' plans are incompatible with the standard even if each was using its respective state-of-the-art abatement technology (like BAT). The only way to meet the standard would then be to reduce the number of pollution sources. This obviously imposes a constraint on economic development. Policymakers can then apply economic instruments, such as taxes or marketable permits, which help to comply with the constraint at minimum cost. Otherwise, determining which of the pollution sources are to be eliminated may be arbitrary and may lead to excessive costs.

Real-world environmental policies usually apply a mixture of technology-based and ambient-quality approaches. One way to operate is to start with the former to obtain an initial assessment of the total pollution in a geographical area assuming that all polluters will follow good abatement practices. If the total pollution happens to stay within the tolerable limits, no steps, apart from enforcing technology-based emission ceilings, have to be taken. Only if it exceeds the limit, the policy maker has to go beyond what is technologically available and to directly constrain the number of sources[xix]. Another way is to start with an ambient quality target, translate it into the total pollution load which can originate from an area, and distribute emission permits so as to not exceed the total. Perhaps, with respect to polluters whose presence in the area is seen as crucial eg, for employment reasons, the permits will take into account available abatement technologies.

7. Typical policy failures

Much progress has been made ever since conscious environmental policies were first implemented. For most of the developed market economies this era started in the 1960s. At the same time, many abatement measures adopted were undoubtedly far from being efficient. Many failed to comply with simple cost-effectiveness criteria. Some policy measures even worsened rather than improved environmental conditions. While the lack of a good theory or expertise available occasionally played a role an important reason for these policy failures was the 'political economy' of the decision-making process.

Environmental policy blunders may be caused by inadequate technical expertise, for instance, wrong toxic levels accepted as a base for setting standards. These then simply reflect the inaccuracy of our scientific knowledge. Perhaps someone had a vested interest to miscalculate environmental risks, or to prevent the academic world from determining a more precise dose-response relationship, but this does not create a systematic bias in assessing what constitutes a good reference point for regulatory decisions. There are illustrations of both under- and overestimated risks from emitting certain chemicals into the environment. Considering freons as perfectly safe gases until as recently as the early 1980s provides the best example of the former. On the contrary, setting the maximum PCB concentration in fish meat in the USA in the 1970s at the level of 2 PPM (Parts Per Million) was probably too stringent when confronted with the toxicological evidence that came available later. It should be noted that the polluters may have vested interests both in underestimating environmental risks, for obvious reasons, and in overestimating them; eg, if they expect they can adapt to stricter standards sooner than their less alert competitors (see Leveque & Nadai, chapter 8 and Gabel & Sinclair-Desgagne, chapter 9 in this volume).

An often quoted example of a policy failing to conform with any cost-effectiveness criteria is the US Clean Air Act as amended in 1977. The amendments required all new electricity plants to install costly desulphurization equipment. This regulation substituted an older one which allowed the plants to meet sulphur pollution targets by any abatement measures such as eg switching to cleaner coal from Western states. As a result of their heavy political lobbying, coal miners from the Appalachia -- who supply the market with high sulphur coal -- succeeded at eliminating all cheaper options and made electricity buyers pay $4.2 billion per year for the higher abatement costs (Howe 1993). Similar failures can be found in many European Union regulations. For instance, the agricultural "set aside" program has turned out not to be as successful as expected. Farmers who got paid for fallowing a part of their land were sometimes found to increase chemical inputs into the cultivated fields thus eliminating environmental benefits of the subsidies.

A policy failure which was very typical in the former centrally planned economies, although in a less transparent form found in other countries as well, is imposing requirements too strict to be taken seriously. The result is not only the lack of compliance, but also a much more devastating long-term effect in the form of undermined authority of environmental administrators and disrespect for the law. With its 32 $\mu g/m^3$ ambient standard for SO_2 (mean annual concentration), Poland exceeds the EU (where the 80-120 $\mu g/m^3$ standards apply), Switzerland, and the USA. Poland is not an exception among the central and eastern European countries, and its SO_2 standard, adopted in the 1980s, is not an isolated case. The Estonian maximum daily concentration of 50 $\mu g/m^3$ is even stricter, as it does not allow for compensating 'dirty' days of eg 80 $\mu g/m^3$ with 'clean' ones of eg 20$\mu g/m^3$; whereas in fact even larger variations in ambient conditions are quite common because of weather variability and other factors. Imposing unrealistic standards is an example of setting policy goals based on what is environmentally desirable without taking into account compliance costs. Of course this may be the result of poor judgement. More likely though, it is a sort of green rhetoric of politicians: an excuse for not adopting any sincere measures to solve the problem.

Some of the best known policy failures have been those caused by perverse incentives inherent in ill-defined property right systems or related to the process of establishing such rights. For decades for instance, in a number of Latin American countries burning or otherwise clearing the forest has been the easiest way to acquire ownership entitlement to the land thus 'developed'[xx]. The mechanism is an important cause of tropical deforestation. Even though some countries, eg Costa Rica, have recently progressed in making the land acquisition process based on more rational criteria, the poverty of small settlers and the vested interests of large corporate holders contribute to the inertia of the old policies.

The case of water rights provides a somewhat less drastic example of how a similar mechanism may lead to the excessive use of a renewable resource. In western United States, but also in some other regions throughout the world, water users need to 'justify' the validity of their permits by keeping water abstractions at a certain level. If their demand for water either permanently or temporarily drops, they have an incentive to keep using the water just as before in order not to lose the permit. This is the so-called 'use it or lose it' principle, a maxim which has been kept stubbornly alive and is utilized by legislators despite repeated criticisms[xxi]. Not only does it encourage wasteful use but it also prevents permits from becoming marketable, and hence creates a double restraint for environmental policies to achieve cost-effectiveness.

Much more subtle varieties of policy failure are those caused by introducing economic incentives without an exact regulatory or behavioural objective. In many countries road transport taxation generates revenues substantially higher than necessary to cover the direct cost of maintaining the road network[xxii]. This suggests that the taxes, to the extent that they reflect environmental costs, can play a role in stimulating a move towards a more environmentally responsible use of cars. A closer scrutiny, however, proves that this is not the case (Button, 1992, pp. 50-51). First, among the least taxed vehicles one can find those which, such as heavy trucks, generate particularly immense environmental and road damages. Second, the taxation generally consists of a sizable fixed component which does not increase with the amount of travel, and therefore incentives to travel less are too week once the vehicle tax has been paid. As a result, transport taxes, however high, are very far from an efficient Pigouvian mechanism.

Even worse than that, in some countries massive government interventions in the transport sector have created perverse incentives to actually increase rather than decrease the amount of traffic. In Germany for instance, those who travel to work by car receive an income tax allowance, which affects the choice between a private car and public transportation. Many cities throughout the world have experimented with direct subsidies for public transportation. The most common result of such experiments is a negligible transfer from private car to public transport and an immense net increase in demand for the latter wiping out any environmental benefits of the modal switch (Button, 1992).

Thus a recent OECD case study of environmental impacts of the transport sector concludes that '(1) prevailing intervention failures lead to an oversupply of transport ...; and (2) a number of these failures induce direct damage to the environment' (Barde & Button, 1990, p.18). In some cases, eg the relative undertaxation of truck transport, the failure is caused by yielding too much to an identifiable interest group (truck companies) or by giving high priority to regional development (believed to be best served by keeping the costs of the freight transport low). In many instances, however, detrimental effects of taxes and subsidies intended as 'environmental', or at least as environmentally harmless, do not emerge because of any targeted lobbying efforts but rather as an unpredictable result of entangled politics of taxation.

Environmental impacts of the transport sector illustrate challenges of so-called policy integration aimed at coordinating various aspects of government interventions. Much research has been done to analyze economic consequences of environmental policies (see Fankhauser and McCoy, chapter 7 in this volume), but the other direction is crucially important too. Policies intended to regulate or support specific sectors, such as transport or agriculture, interfere or simply undermine environmental policies thus imposing on these additional (sometimes unexpected) constraints.

Examples such as those referred to in this section make clear that environmental policies, whose important purpose is the correct market failures, fail themselves quite often too. Sometimes this failure is rooted in politics and could have been avoided had the government or environmental authorities found a feasible mechanism for compensating potential losers. Sometimes correcting the failure does not demand a political compromise but requires simply a better analysis and more insight into the system to be regulated. In any event, studying the economic aspects of alternative policies can assist in identifying and dealing with constraints these policies inevitably face.

8. Do economists contribute to environmental policy debates?

The purpose of this chapter was to report on how economists approach environmental policy, and what constraints they are confronted with. There are three major concepts which economic analysis applies to assess policies and to recommend specific measures: effectiveness, efficiency, and equity. The traditional focus was efficiency, or a somewhat weaker criterion of cost-effectiveness. Only in the late 1980s did economists notice that the real-world policymakers largely ignored their recommendations. For instance, surveys show that the scope of use of economic instruments has been much smaller than theoretically envisaged. What makes the economists' advice so little practical?

First of all, one must admit that, despite the widespread disappointment among economists, there are a number of successful applications of economic approaches to environmental policy. In the United States marketable pollution permits lowered abatement costs by thousands of millions of dollars. In Europe, deposit-refund systems helped to deal with the plague of wasteful packaging, and environmental charges, even at relatively low levels, did provide incentives for improvement. Some charges eg, the Dutch water pollution charge, have had significant impacts on emissions and their abatement. In economies in transition, recovery programs crucially depend on the operation of environmental funds originating from pollution charges. These are just a few examples of when economists' advice has been taken into consideration.

However, it should be observed that economists' early interests largely missed out on what was the focus of environmental activists and policymakers. While the former were preoccupied with effectiveness, the latter seemed to be overwhelmed with equity questions, as well as administrative practicality of alternative solutions. Not many economists were ready to adopt a pragmatic approach, to study dilemmas confronting politicians, and to provide them with politically viable recommendations.

A distinction between the traditional economic approach and the one which incorporates various concerns voiced by non-economists has often been compared with the division between what is called environmental economics, and ecological economics[xxiii]. While past experience may justify linking the former with a single analytical method, ie, that of 'neoclassical economics' and game theory, and the latter with the sustainability concept, most recent developments indicate that both fields largely overlap now. One can observe 'neoclassical' and game-theoretic analyses of complex ecological-economic problems which seemed to be beyond the domain of economics before.

What do economists contribute to environmental policy debates? First they assist policymakers in assessing costs and benefits of alternative measures or projects to determine whether these are recommendable on economic grounds. Second, they advise on the available instruments to carry out a given policy, and suggest those which are most likely to bridge the gap between common-sense expectations and economic efficiency. Third, they pay attention to indirect effects of alternative policies which are often overlooked by non-economists; for that reason economic models can be important and revealing.

Various objectives are articulated and promoted by various interest groups. However, there is no natural constituency for macroeconomic efficiency, since the effects usually are distributed among wide social strata, and no particular group directly gains from it. It is here where economists play a stimulating role as a surrogate interest group pushing for solutions which minimize overall costs to be borne by society.

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ii. There is also a less technical and more political notion of equity which refers to providing citizens with services they 'deserve', and charging them what they 'can afford', but it will not be used in that sense in this chapter.

iii. See Pezzey (1989) for a thorough review and analysis of many definitions introduced in professional literature and politics language.

iv. See Buchholz (1997) for an elementary exposition of an intergenerational equity rationale and criteria.

v. The materials balance approach, initiated by Kneese et al (1970) and followed by very few economists, is exceptional in its emphasis on studying the physical volume of matter flowing through economy. The approach keeps trace of all the material inputs (including those raw materials which are not purchased, such as air), and outputs (including those which are not sold, such as waste products) of economic activities. It indicates that what commercial transactions capture and conventional economic analyses deal with is but a fraction of the flow which determines production and consumption, as well as the characteristics of the environment we live in. See Ayres (1998) for a recent exposition of the materials balance approach.

vi. See Tahvonen (chapter 18 in this volume) for explanation of the 'profit maximizing sustained yield' concept.

vii. Young (1992) derives more than 30 practical 'prescriptions' from the sustainability concept, addressing more directly issues public policy has to deal with.

viii. Weitzman (1998) carried out a large survey of economists asking them what real (ie net of inflation) interest rate would be most appropriate for calculating NPV of environmental projects with very long time horizon, such as measures to mitigate climate change. The rate most commonly argued for was 4%.

ix. Proofs of these theorems require standard microeconomic assumptions on u(c,s), such as concavity, separability, ie $u(c,s)=u_1(c)+u_2(s)$, and monotonicity of u_1 and u_2. We refer readers to Beltratti et al. (1998) for further technicalities and bibliographic details.

x. If it does ie, if it creates an unfair advantage over competitors from other countries, an ecological dumping takes place. See Rauscher (1992) for a deeper analysis of this problem; subsidies can be substituted by more liberal standards of what the polluter is responsible for.

xi. The European Union was more specific in defining the terms of a 'transition'. Initially a six-year period was envisaged (1974-1980) with the maximum subsidy rates of 45%, 30%, and 15%.

xii. This paragraph and the next are based on unpublished material written by Jean-Philippe Barde.

xiii. Economists study the so-called tax incidence to see how the burden of an imposed payment or regulation is divided among producers (through reduced profits) and consumers
(through reduced real purchasing power). In general, the incidence depends on the shape

of supply and demand curves rather than on intentions of taxing authorities. In particular, it does not depend on whether the seller or the buyer are technically made responsible for transferring the tax to the Treasury.

xiv. This is the idea widely known as the Coase theorem; see Löfgren (chapter 1 in this volume). Readers should be aware that this theorem requires a number of formal assumptions which are quite restrictive.

xv. It has been recognized, for instance, that in Europe trying to achieve substantial CO_2 emission reductions at minimum cost would require that western European countries compensate the rest of the continent for its abatement effort beyond what the East is prepared to undertake as part of its domestic policies (Bohm & Larsen, 1994).

xvi. In the United States the concept of 'federalism' has been studied which is equivalent to the European Subsidiarity Principle.

xvii. See Barde (chapter 6 in this volume) for an illustration of the welfare loss caused by setting standards that are computed as "average" for several locations.

xviii. It has been established that long-term exposure to the concentrations of sulphur dioxide as low as 20 $\mu g/m^3$ inhibits growth and can be harmful. In contrast, these concentrations can be twice as high before they lead to evident adverse health effects.

xix. This can be done in a number of ways such as total bans on development projects, establishing zones for specific activities, regulating the level of production or simply distributing pollution permits in a stricter way than suggested by technological considerations.

xx. In Brazil a very complicated system of public land allocation and solving land disputes encourages clearings larger than what is used for subsistence or commercial purposes. By clearing the land one not only claims the entitlement but also prevents squatters from invading the economically useful 'core' of one's estate (Binswanger 1989).

xxi. The continuous use of a water-discharge or air-pollution permit is a prerequisite for keeping it under the Polish environmental law which, however, after 1989 was reformed towards greater effectiveness and efficiency. It is amazing how persistently the lawmakers insisted on including the 'use it or lose it' clauses in all the drafts of new legal acts.

xxii. In the Netherlands they cover 434% of road expenditures, in the United Kingdom 335%, in New Zealand 235%, in Sweden 230%, and in Denmark 214%. In some other countries they cover between 100% and 200% of such expenditures. See Button (1992) p.49.

xxiii. Both orientations have their academic societies: Association of Environmental and Resource Economists (mostly in the USA) and European Association of Environmental and Resource Economists, and International Society for Ecological Economics, respectively. Both of them have their journals: Journal of Environmental Economics and Management (the publication of the US association), Environmental and Resource Economics, and Ecological Economics, respectively.