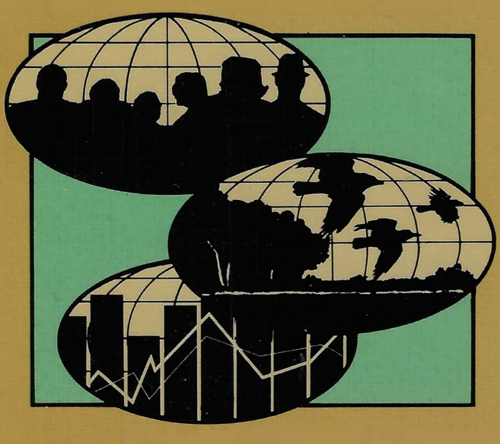
Macro-environmental policy: principles and design

Huppes, G.

DEVELOPMENTS IN ENVIRONMENTAL ECONOMICS - 3

MACRO-ENVIRONMENTAL POLICY: PRINCIPLES AND DESIGN

GJALT HUPPES



ELSEVIER

To the reader

This pdf book is a scan of the original hardcover version, which has been out of sale for quite some time now. Scans are not perfect, so there are anomalies in several pages. Though three decades old by now, the subjects are awkwardly contemporary.

The full contents and the contents per chapter have hyperlinks to the corresponding pages, for fast-and-focused reading. At the time all had been hand-typed.

Enjoy,

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Amsterdam, 22 April 2022
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DEVELOPMENTS IN ENVIRONMENTAL ECONOMICS

VOLUME 3

Macro-Environmental Policy: Principles and Design

by

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PREFACE

This book is the result of an extended PhD study. It owes much to research during the last decade at the Centre of Environmental Science of Leiden University (CML), especially in the *Substances & Products Section*. Most of that research has been financed by external contracts with some firms and with many public organisations in the Netherlands and the EC. The book attempts to develop a set of policy instruments which have as common factor that they exert a very broad influence on environmentally relevant decisions in society but in an indirect way only, by changing their context. Analogue to the main distinction in economic policy, this is the domain of policy instruments for macro-environmental policy, while current environmental policies are mainly at a micro and meso level.

I hope the book may be of use to universities, both in economic, administrative and environmental departments, and to governments, in their strategic development of policy instruments. Some more technical parts on specific instruments, especially on the methods for life cycle analysis, may be of interest for firms and consultants in that field and for NGOs. Although some parts have an abstract or specialised nature, the book as a whole can be understood by generalists.

The general line of reasoning owes much to the basic ideas in macro-economic policy where instrument variables, through modelled intermediate variables, influence a number of target variables as the aims of the policy. Tinbergen's book on that subject, a source of inspiration, has also fathered the title of this book. It is my conviction that, without such macro-instruments, environmental policy will either become insufficient environmentally (if it is not so already) or extremely costly in both an economic and a social sense.

New in this book is the general framework for development and evaluation of instruments for macro-environmental policy. The details as filled in will be disputable on normative grounds, an unavoidable state of affairs in the realm of politics. This is the case especially in the choices on normative principles that define the aims of the instruments. Also, the lines of reasoning, the structuring of submodels, and the few empirical assessments made all have a preliminary status, still with many unclear and too personal elements in them. Further scientific contributions are necessary in many of these fields. Also new, at a detailed level, is, firstly, that the main financial instrument, the "Baumol-Oates emission tax," can be implemented at a macro level, for society as whole, a novel instrument, the substance deposit. This seems the case at least for several substances that are important in current discussions on environmental policy, such as carbon, heavy metals, acids and phosphates. Secondly, I propose a solution for one main problem in environmental life cycle analysis, that of the allocation of environmental effects caused by a multiple economic process to each of its services or products.

Research projects in the Substances and Products Section at CML have been of a collective nature, without exception, so in a sense all former and current colleagues contributed to this book. Some colleagues contributed more directly to specific parts: Paul Mulder, Ruben Huele, Jeroen Guinée, Ester van der Voet and René Kleijn. I received technical support from our CML librarians Edith de Roos and Jetty Staats, from Peter de Putter, who helped to expand and put in order my literature collection, from Arnold Cohen who helped to correct my imperfect English style, and from Henk Bezemer and Bert Snoek, who helped to meet the publisher's deadline. The Cosijns and the Nusts supplied a quiet place to work in their homes, where I spent much time separated from my wife and children.

The formalities of the thesis preparation were not formalities at all. Helias Udo de Haes, my supervisor, helped to structure the work, in several rounds of discussion, as did Bob Kagan, my second supervisor, though mainly from a distance. Comments by Jan Pen and Wim Hafkamp, the formal referees, and by Robert van der Veen led to substantial changes.

The book has been my job and is my responsibility. It reflects what I think is important in my work as an environmental scientist.

Amsterdam, 31 July 1993
G.H.

CONTENTS

PART 1 GENERAL INTRODUCTION

1.1 INTRODUCTION TO MACRO-ENVIRONMENTAL POLICY

During the last decades the volume of environmental regulation has been expanding in all Western countries and is starting to do so all over the world. This development is taking place while production and consumption grow several percent per year. Human activities increasingly affect the global ecosphere, with new tasks for environmental policy steadily emerging. Climate change and ozone layer depletion have materialised as problems that need to be solved. The total eradication of major habitats is under way, especially those of forests in the tropics, diminishing species diversity at a global level. Added to this is the selective destruction of species such as elephants and rhinos for their ivory and horn and the trading of numerous species alive. Some older problems related to hazardous emissions are under control. Many waterways are cleaner now in terms of their loads of oxygen demanding substances and heavy metals than two or three decades ago. It would appear, however, that this improvement has taken place at least partly through problem shifting. The amounts of heavy metals mined have increased substantially in the same period. They are now disposed of in other ways or they are trapped in products such as building materials, for later disposal. Problems such as acidification and eutrophication are becoming controlled but at a level where long term effects may still seriously damage the quality of the environment.

There are at least six reasons for a re-assessment of the instruments for environmental policy, as related to its increasing intensity, complexity, scope and domain. Growing production and consumption exert a rising influence on all environmental problems. At the same time, the amenity value of the environment becomes increasingly important to a population that is increasingly prosperous in terms of material welfare. For these two reasons the regulative task of environmental policy has already increased very much, as indicated above. However, welfare growth, in market terms, requires more than regulative activity because of the mass of environmental effects and the higher valuation of environmental quality. In addition, welfare growth involves a rapidly increasing number of different processes, materials and products. This is another independent factor increasing the burden of regulation. If no disasters occur, these three factors - volume growth, a higher valuation of environmental quality, and differentiation in products and processes - will remain operant for a long time to come.

It is not only the number of items to be regulated and the strictness of these regulations that increases because of the factors stated, but the complexity of regulation itself. The higher number of problem domains of environmental policy requires an increasingly complex internal and external coordination in policy formation and implementation. The desulfurization of the rapidly increasing use of sulfur-containing energy resources is an example. Using current technologies, one tonne of sulfurdioxide abatement (in kg of sulfur) requires the equivalent of ten tonnes of coal, a depletable resource, with many environmental effects at the mining phase, such as substantial emissions of the climate-changing gas methane. It also requires over three tonnes of limestone, a depletable resource, with local habitat destruction as a result. It produces eight tonnes of carbon emissions affecting climate, and also toxic wastes including several kilograms of heavy metals from the burning of the coal required. The burning of the coal leads to sulfur emissions of around a third of a tonne if they are not desulfurized themselves. The transformation of limestone into gypsum, by binding sulfur, frees and additional amount of carbon dioxide, around a third of a tonne (in kg carbon). Finally, it produces a quite contaminated form of gypsum, partly in products and partly as a waste. Developing consistent rules here becomes a more complex task. More complex systems of rules are more complex to implement, comply with, and enforce - e.g. through policing. These complexities alone are reason enough to re-assess the approaches to policy instrumentation that have developed in the last decades.

Increasing economic activities have produced ever more environmental problems. Environmental policy started with local problems, in the nineteenth century. The regional problems of smog forming, acidification and eutrophication led to the upsurge in environmental policy in the Seventies. The global problems of ozone layer depletion and climate change have only been receiving scientific and political attention in the last decades and they will certainly be followed by others. Improvements in terms of one problem will usually lead to negative effects to others. The increasing scope of environmental policy thus leads to increasing problems of coordination in policy formation and implementation, a fifth reason to take a fresh look at policy instruments.

Environmental policy, however, is now turning to prevention, the sixth main reason to start searching for other instruments in environmental policy. There is a growing awareness that a major shift in environmental policy is required to ensure that the effectiveness of regulation keeps pace with the autonomous developments of population growth and increases in material welfare. A shift from mainly end-of-pipe techniques to prevention is required for quantitatively adequate results. If wastes are increasingly contaminated and their amounts increase, the effectiveness of end-of-pipe cleaning technologies has to improve at their combined rate of growth. Preventing some types of wastes and diminishing their amounts through the use of other processes and products and through their redesign, often may be cheaper and more effective then applying increasingly complex cleaning technologies that themselves cause other problems. The trend towards prevention is becoming accepted politically throughout the Western World.

With prevention, the domain of policy broadens to all economic decisions in production and consumption, as each decision will be relevant in terms of environmental regulation. The domain of policy is no longer restricted to where the evil occurs, but extends to the activities that, indirectly, cause the evil. My household cleaning habits, harmless by themselves, lead to problems in sewer purification. A personal computer requires a certain amount of CFCs in its production. Curbing their sales or changing their composition may thus be worthwhile environmentally. Both the domain of environmental policy and its complexity increase substantially with the application of preventive regulative measures. Assessing the environmental reasonableness of some preventive measure, such as forbidding the use of plastic bags that are harmless in their use, requires a knowledge of other processes in production, consumption and disposal than the one regulated. Increasingly, these processes themselves are already subject to policies that cannot be abandoned without a loss in environmental effectiveness. This worrisome increase in complexity is no reason to dismiss prevention. The basic reasoning behind it remains valid: prevention may really help diminish environmental problems, at costs that may be substantially lower than those of increasingly complex end-of-pipe technologies. Again, there is a quantitative administrative problem involved of more, and more complex, regulations. There is the additional problem, however, of how the analysis for prevention is to be made. Analysis for prevention is such a complicated affair because indirect, or secondary, effects of regulations, through induced changes in other products and other processes, are the prime motive for such regulations. In current regulations, such indirect effects usually have been neglected, under the assumption that the prime direct effect is more important than all indirect effects together. With current policy instruments also being used for prevention, the analysis required can never be that simple again.

An old fashioned health hazard such as cadmium[1] may illustrate the type of problems one encounters in this broader analysis of indirect effects required for prevention. Cadmium causes severe health problems in higher organisms. In many places, its diffuse concentration in soils is building up to levels that will make them unfit for food production within decades. It is extracted from the substrate we live on, mainly in zinc ore and, in smaller amounts, phosphate ore. Iron ore, copper ore and fossil energy are much smaller sources. The cadmium mined is partly stored in wastes and partly emitted, both in mining and ore processing and in the combustion of coal and oil. All remaining cadmium penetrates further into the economy, both as a contaminant in metals and as a valued metal itself. For technical reasons but increasingly for environmental reasons as well, zinc and other metals should be virtually cadmium free. The amount of cadmium produced as a fraction of the cadmium mined in zinc ore is thus increasing, as is the amount of zinc[2]. Even so, the market price of cadmium has increased over twenty times between 1985 and 1992. It now is a very expensive metal, with a price of over $10 per kilogramme. To prevent emissions, some applications of cadmium in products have been restricted or forbidden, e.g. as a bright yellow and green dye and as a stabilizer in plastics. Its use in rechargeable batteries is soaring however, that is the main cause for its recent price rise. Should this application be forbidden as well? Forbidding some application of cadmium is extremely effective directly. If cadmium-containing dyes are no longer used, the cadmium emissions from discarding coloured products will stop. If there is no cadmium in batteries, the cadmium emissions from batteries will stop. Would these be sensible policy measures, from an environmental point of view? Based on the old fashioned direct analysis the answer is unequivocally 'yes'. The indirect-effect analysis, or second-order analysis, that is required for prevention may complicate the picture. Commercially used cadmium, being a by-product, is very inelastic in its supply. This means that a certain amount of cadmium will be produced almost regardless of its market price. Forbidding some applications will lead to an increase in others as the price drops. This is one first example of indirect-effect analysis. Depending on the ways the several cadmium flows affected are treated as a waste, this second order analysis of policy measures may show a net improvement, or deterioration.

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[1] See the case on cadmium below, in part five, for a fuller analysis.

[2] A new, more energy efficient process for zinc ore processing has been developed that fully cleans the zinc and all waste from cadmium. With this process, virtually all cadmium is extracted in metallic form, as a marketable product.

The indirect-effect analysis may be carried one step further. Suppose, for the sake of argument, that the net effect of cadmium policies is zero in terms of their effects on cadmium emissions. The products formerly made with cadmium are now partly replaced by others, e.g. other dyes, other stabilizers, and other batteries. Each of these has its own associated smaller or larger environmental problems unrelated to cadmium. Considering the indirect effects of these other dyes, stabilizers and batteries surely would make negative the net effect of the cadmium policy that started all these changes. The indirect-effect analysis may be taken one step further again. As direct equivalents to cadmium-containing products tend to become more expensive and less attractive, people will change their spending habits from cadmium-containing products to other luxury items such as holidays abroad, by car or plane, and many other activities. The net balance - after trading the effects in some way of different environmental problems such as potential health effects, acidification and global warming - would certainly become negative. The conclusion here is somewhat irksome. If the indirect-effect analysis stops at the cadmium effects of substitution, indicating only the shifts of cadmium applications caused, it clearly is inadequate. Important aspects caused by the problem shifting then remain unnoticed. If it is to include all relevant effects on all other problems affected, the analysis is clearly extremely complex, if not impossible. Prevention, as an important ingredient in future environmental policy, needs an adequate type of indirect-effect analysis if specific measures for prevention are to be argued. Or other types of policy instruments should somehow solve this problem of analysis.

The central assumption here is that the main instruments of environmental policy, individual permits and general technology standards, will run into serious trouble when applied to the expanding tasks ahead. This situation is the result of the sheer volume of the increasingly complex tasks and the lack of a mode for the indirect-effect analysis and assessment of effects, which makes them unfit for preventive policies. The main question is how policy instruments could apply to a smaller number of decision-making units in society in a simpler manner, while being increasingly effective overall. Other policy instruments are sought in all Western countries, especially financial instruments, such as taxes and subsidies, and informational instruments, such as audits and ecolabels. The most promising direction for the development of such new instruments is that which abandons the direct regulation of behaviour and ever-larger amounts of specific products and processes. The ideal instruments should influence larger units of society in a more indirect manner, applying to all ranges of preventive measures and end-of-pipe technologies alike. An analogy to economic policy can be made here. Instead of governments themselves starting new firms, developing new technologies and opening up new markets as the means to increase total income and employment, governments have chosen to use indirect instruments, such as those of macro-economic policy, to realize these aims. Instruments for macro-economic policy - such as the interest rate, the total amount of spending and the exchange rate - influence decisions at the micro-level without the policy-makers ever knowing which decisions are changed, let alone that they would interfere with such decisions in any direct way. The behavioral changes resulting lead to the desired aim statistically, as the final result of partly stochastic processes. As in economic policy, environmental policy could use indirect instruments for macro-environmental policy, i.e. *macro-instruments*. The central aim of this study is to contribute to the development of instruments for macro-environmental policy and to assess their practical applicability.

The term 'policy instrument' is used in a narrow sense here. Planning is often referred to as an instrument, e.g. environmental policy plans. Such plans are instruments, but instruments mainly to guide or bind the behaviour of governmental organizations, including the plan-making unit itself. Such instruments are not included in the definition of policy instrument here[1]. There are also instruments that may be used by firms to adjust their behaviour according to environmental criteria. A software programme that helps farmers to minimize their nutrient losses is such an instrument. Such privately produced instruments that change the behaviour of other private agents are not policy instruments in the sense used here either, however relevant they may be from an environmental point of view. The restricted meaning adhered to is that policy instruments consist only of regular public activities aimed at direct changes in the behaviour of individuals and groups in society.

Developing policy instruments is a design process that, such as any other design process, is guided by aims and principles (the normative part), and by some knowledge of the reality the design will funcction in (the empirical part). Practically, the design process does not start from scratch. It starts with examining existing designs both as a basis from which to work and for inspiration. Each of these three elements - aims, models, starting points - guides the design process. Choices of aims, models and starting points cannot be fully established themselves on firm foundations. They are necessary, however, to make the subsequent design process a rational one. The subjective positions taken here can best be described in terms of these three elements. Of course, the factual process of design has been much more irrational. The chance events come to mind of educationally-bound research and contract research for governments and industries. In this study the loose ends collected over the course of time are knit together, as a rational reconstruction.

The aims and principles of environmental policy instruments might be formulated politically, as the programme of some green party. The arguments then may involve notions that are accepted there, such as the primacy of environmental and distributional concerns, as opposed to growth and technological progress. This study, however, strives for a higher level of generality, requiring an analysis of the extent to which such principles as freedom, justice and equity relate to the choice of instruments of environmental policy. In this way the results, as far as dependent on the notions developed, should be acceptable to the broad political spectrum that has developed in the tradition of the Enlightenment, from conservative liberals to progressive liberals and socio-democrats, including most greens.

Choices on aims and principles are not the only normative choices made. The restrictions of the empirical model chosen imply normative choices as well, at the level of its main structure. First, in this model, there is a clear and absolute distinction between government and society. In real political life there may be interest-bound political processes that lead to certain decisions and not to others. However, the model does not include elements of the political process as an interplay between government and society. Rather, it supposes that, ultimately, policy choices, such as that of the instruments for

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[1] In France, the *plan indicative* as developed in the Sixties, not only was to guide public activities related to the economy, but also private actions. Businesses were advised to adjust amounts to the plan to prevent any shortages and surplusses. Only the latter aspect would fall under the strict definition of policy instrument used here.

Rather, it supposes that, ultimately, policy choices, such as that of the instruments for environmental policy, are based on rational arguments only, related to the expected effects of the application of the instruments as specified. This study is about rational arguments at the collective level and does not treat any aspects of the political process that may act as restrictions, except through the arguments as developed. The position taken thus reflects that of a benevolent dictator or a humble citizen trying to make up their mind regardless of specific effects on their personal situation.
Secondly, the empirical model posits a related restriction on the type of policy formation allowable. Policy formation and policy implementation may be intricately mixed, of course, as reality proves. The causal flow as described in the model here goes only one way, however, from government decision, through implementation, to society. Thus the model chosen cannot reflect the type of (semi/neo)corporatist policy instruments that are favoured by some, such as those advocating 'horizontal government'. In horizontal government, policies are reformulated and reshaped during implementation, or they are even shaped in that same process.
Thirdly, the empirical model pictures the environment as an independent system on which negative effects now threatening it are to be avoided as much as possible. No attention is paid to the possibility of positive action directed towards the environment itself, designed to change basic environmental processes. Solutions to the global warming problem, for example, are represented in the model only in the use of instruments to reduce the emissions of climate-changing substances. Of course other measures are possible. One could, for example, throw trace nutrients into the oceans to absorb large amounts of carbon dioxide through increased algal growth, or could shoot particles into the upper atmosphere to restrict the incoming flows of solar energy, see for example Nordhaus (1991). Similarly, the genetic diversity in the environment might be preserved effectively in gene banks, with only some individuals taking part in actual reproduction under controlled conditions. Or the quality of forests could be increased by pest control instead of diminishing acid rain as a negative health factor, see Skelly (1992). According to the position taken here, these methods may be applied only if limiting negative effects of society on the environment proves impossible. The arguments for this choice, not worked out here themselves, relate to the stability of the global ecosystem and the flexibility of the human use that can be made of it[1]. This stance on the independence of the environment is quite debatable.

One starting point, for inspiration, has been economic policy. There, the last two centuries have shown how highly effective indirect policy instruments can be. From Adam Smith, with his plea for the institutionalization of markets[2], to Keynes in his proposals for macro-economic policy, there has been a succession of theories and related proposals that have in common that direct regulations are means of last resort, e.g. for wartime use, with indirect, more macro regulations as the main instruments to foster economic welfare. Two authors from the Netherlands have been chosen, representing two

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[1] A practical argument against lessening this restriction in the model is that very awkward technical interdependences are avoided. Restricting incoming solar radiation, for example, will lower the energy proceeds of solar cells and will increase fossil fuel consumption. Such interdependences would seriously hamper effective policy formation.

[2] Smith gives as an example how taking grain from rich traders and peasants to distribute it among the starving poor, although releaving hunger temporary, would result, indirectly, in a diminished grain production, with more poor people starving.

streams within this greater economic main stream. One is Tinbergen (1967), with a survey of the practical techniques of macro-economic policies and their place in society. The other is Zijlstra (1966, with Goudzwaard), filling in the gap between direct intervention in the economy and the highly abstract categories of policy instruments as depicted by Tinbergen. The title of this study is based on that of Tinbergen's 1967 book on macro-economic policy.

The other starting point is the literature on policy instruments as found in many disciplines. A complete survey is quite impossible, so choices may sometimes be seen as arbitrary, and to some extent they are. However, a broad spectrum has been covered. The economic literature on different instruments in particular has been covered quite extensively.

With these ingredients the study now proceeds towards the aim stated. To help the reader find his way, some remarks now follow on the function of the different parts and chapters.

1.2 A READER'S GUIDE

The study has been divided into four main parts. The following three parts, together comprising half the book, develop the theory and the design of macro-instruments. The fourth part, also roughly half the book, contains four case studies. They show how several macro-instruments together might work in specific situations. Their function is first, to indicate if the macro-instruments developed are applicable. Their second function is how they may work out in the quite dissimilar domains of economic activities chosen as cases. The case studies are not intended to throw new empirical light on the problem areas treated. They include some deviating positions, however, to keep the mind of the reader alert. These four main parts are embedded between this introduction and a concluding and summarizing part. The latter particularly examines the possibilities for a changeover to more macro-instruments. The summary allows a rapid survey of all subjects. The reader may skip any of the theoretical or empirical sections and limit his reading to its summary.

Following this Part One, Part Two first builds the analytical framework for the empirical analysis of policy instruments. The main problem there is how the general lack of widely accepted knowledge on societal and environmental processes can be circumvented, while still leaving essential ingredients for defining instruments. The more detailed further empirical analysis, as required for quantified application, may then be shifted to the application of instruments. The instruments as described in the literature are then confronted with this system of classification, and they are fitted in, after some adjustments, or omitted as not relevant to the design problem to be solved. The result is a survey of instruments, ordered as to type of process and, per type, according to their capacity for aggregate applicaiton, i.e. how macro they are. Two of the macro-instruments, the most macro- economic and the most macro-cultural, are especially important. They are worked out in Part Four.

Part Three begins by constructing the framework for the normative analysis of instruments. With it individual instruments are first ordered. This ordering is then used to develop a strategy for policy design called the *flexible response strategy*. Policy design is one level more comprehensive than instrument design; it may use several instruments as building blocks. The strategy states the order in which instruments are to be used preferentially, until the problems are solved acceptably.

Part Four works out two main options for macro-instruments in more detail. The first option is a *public system for the life cycle analysis of products*, the most macro-cultural instrument. After a short general survey on the development of this instrument some specific contributions are described. The emphasis is on how to structure the empirical analysis of indirect effects, and on the requirements and procedures for evaluating product alternatives. The second option is the *substance deposit*, the financial instrument that is most macro in character. It is based directly on the substance flow analysis of the economy, a subject that will be summarized shortly. Some variants remain open and their pros and cons will be discussed.

Part Five gives an indication of how the flexible response strategy may work out in four different situations. One is the situation of the marketing of a new product-material combination, the polycarbonate milk bottle. The life cycle analysis of products plays a central role in this actual Dutch case.
The second case is on cadmium, as a main representative of the group of toxic heavy metals. The substance flow specified is that for Europe. The flexible response strategy is then specified as to applicability, the instruments are quantified to a degree, and a provisional assessment of its effectiveness is made.
The third case is on nitrogen and phosphorus as main eutrophicating and, the case of nitrogen compounds, also acidifying substances. As with cadmium, the flows specified are for Europe. A specification and assessment of the flexible response strategy follows.
The fourth case is of a mixed nature. It deals with energy depletion and global warming, the connection being the use of carbon, resulting in the emissions of carbon dioxide and methane. A separate analysis in an appendix, which comments on a main Dutch proposal for energy taxes, concludes that there is no depletion problem related to carbon containing substances, only an emissions problem, e.g., that of global warming. The by-now-familiar procedure then starts, beginning with the substance flow analysis of climate changing carbon compounds, this time at the global level, followed by the specification of the flexible response strategy and an assessment of its effectiveness.

To conclude, Part Six, first summarises what has been produced in the study at a a theoretical level, converging in the flexible response strategy and how this strategy might be applied in the cases. Then the assessment follows of the more general applicability of the flexible response strategy, at several administrative levels. The study ends speculatively, with the short and long term prospects for the realization of that strategy at various administrative levels.

PART 2 FRAMEWORK
A GENERAL FRAMEWORK FOR ANALYSIS

2.1 INTRODUCTION TO THE GENERAL FRAMEWORK

As the cadmium example has shown, society is too complex for a "full" analysis of all indirect effects of policies. Even assessing effects of established policy changes, ex post, usually requires modelling of a very complex and often debatable nature. The model would still have to predict what would have happened if some other (non)policy had been chosen. Careful empirical research may give reasonably ascertained facts, one might think optimistically. This is the case only in rare exceptions and even then the causal analysis remains debatable. One such example is Bressers (1983; 1988). In a very meticulous policy analysis Bressers gathered the data on the Dutch Law on the Pollution of Surface Waters from 1970. In that law, a substantial tax[1] was introduced on the effluents of all larger firms, to both sewer and surface water. Payments were, and still are, differentiated according to the amounts and composition of the waste water. Oxygen demand is a main factor in the tax level but heavy metals are included as well. In contrast, small firms and households pay a flat rate, regardless of the amount of effluent and its composition. The proceeds of the tax were used to cover the costs of collective waste water purification. These costs include the taxes these purification bodies themselves pay for their discharges of purified effluent into state waters. At the same time a system of permits was introduced, limiting the amounts of several substances in company effluents. The implementation of the tax system and of the permits was the responsibility of the Dutch Water Boards, institutions dating from the Middle Ages. The effluent amounts of larger firms dropped by over 90% percent in the next fifteen years, reversing a steeply rising trend in the preceding years, see Huppes and Kagan (1989). The remaining ten percent are purified collectively, nearly nullifying the formerly dominant emissions of larger firms. No reduction has taken place in the amounts emitted by smaller firms and households.

How has this drastic behavioral change been brought about? Several mechanisms may have been responsible, singly or combined. First, consider the explanation given to Bressers by the officials of the Water Boards. They perceived that their careful analysis of possibilities for emission reduction, and their intimate talks with officials of the firms concerned, induced the firms to redress their formerly bad ways. The new installations were fixed in permits. The Water Board remained active and spotted new possibilities for emission reduction. Techniques were often introduced by the firms which resulted in substantially lower emissions than those allowed in the permits.
Secondly, Bressers also analyzed the data statistically. The decentralized nature of the nearly twenty local Water Boards led to different tax levels in different areas. This gave him the unique opportunity to analyze the effects of different levels of the same kind of tax, introduced at the same time, in the small and culturally quite homogeneous country of the Netherlands. His findings that these differences in tax levels explain to an astonishing degree differences in effluent reductions of the larger firms, suggest that the diligence of the Water Boards was not as significant a factor.
There are alternative explanations, however, to account for the overall reduction that occurred. During the period studied energy prices rose four to six fold, from around $5

[1] Throughout this text 'taxes' is used without indication of a specific judicial status. 'Tax' thus is synonimous with 'levies'.

per barrel of crude oil to a brief peak of nearly $50. Many of the emission-reducing techniques also saved energy. Was that the crucial factor? And of course in the same period firms became generally aware of their responsibilities in environmental affairs. How much of the behavioral change would have been induced by each of these factors alone? Such general questions are extremely difficult to answer convincingly. In any case, it is clear that such broad changes in prices and attitudes lead to substantial behavioral changes. These alternative factors cannot explain differences in emission reductions between the groups as stated.

Bressers was in an exceptional situation where statistical analysis could ascertain "the facts" quite reasonably. His extensive study thus is a very unique exception in *ex post* policy analysis. Even in this case the suggested cause of the emission reduction - the effluent tax - remains open to dispute. In most *ex post* analyses, there are no facts that allow such a relatively clear causal analysis of the changes induced by certain measures. As in dynamic situations many changes go together; disentangling their respective contributions to emission reduction remains a highly hypothetical affair. Usually only limited opportunities exist to restrict the domain of possible alternatives by hard facts. Any assessment more precise than a statement of all possible causation will thus be based on more general, theoretical notions of how policy implementation works, how society functions, and how technologies influence the environment. Thus, *ex post* analyses with clear results can generally be based only on the acceptance of general notions of social processes.

The *ex ante* analysis required for policy design depends even more on such general notions. In this case one does not know yet what exactly the future will bring; that future itself must be predicted, as one possible future. The choice of policies then is based on comparing several possible futures resulting from several corresponding policies. The general notions on relevant chains of cause and effect may be separate bits. "If you hit them hard they're sure to run" may express a certain group of ideas on how to correct imperfections of the market mechanism leading to environmental problems. As a strategic policy approach it would fail. In most Western societies, and certainly in Holland, it is not the done thing to hit citizens, or firms, hard, unless their behaviour is truly exceptional. If most farmers emit too much nutrients, a single farmer cannot be exceptional. He simply will not be hit and no environmental effect will occur. See Huppes and Kagan (1989) on a typical example of non-implemented, non-effective Dutch legislation to restrict intensive animal husbandry. It does not seem sensible to base strategic policy design on such loose notions, where outcomes depend on the loose bits chosen. A more systematic approach is desirable that, in the end, links different designs to their respective outcomes. This approach makes possible a more rational development and choice of policies. Full modelling of all environmentally relevant processes being impossible, the best course is to set up a main model structure in which a number of "loose bits" can find their place systematically. That is what this part of a framework for analysis is about.

The social nature of the facts involved makes this modelling a less-than neutral affair. The model chosen will favour one policy option as more effective than another. The choice of model is subjective, so what does it mean if someone proposes a model? Is it merely his subjectivity that is reflected? Luckily, some notions are accepted quite widely.

Government actions against private persons and institutions are based on laws or citizens will not accept them. The efficient use of resources in production requires the functioning of a number of markets. Markets require the definition and safeguarding of property rights. The individual procurement of collective goods will lead to a smaller amount of collective good than any single individual would want. It seems reasonable to make such general notions on how society functions the basis of models expressing the effect chain of environmental policies.

If models for environmental policy only state the expected effectiveness of instruments in terms of the environmental aims stated, they still have no practical value. The parameter of environmental effectiveness alone is not enough when choosing between instruments for environmental policy. Comparisons are to take into account effects on other variables as well. If two policy instruments or sets of policy instruments would have exactly the same environmental effects, the choice between them can only be based on other types of valued effects resulting. Thus, models will be more complicated, indicating other valued effects as well. Main values and aims to be incorporated in the evaluation of instruments are discussed and selected in Part Three.

It is not only separate effect chains that are to be modelled here. One specific aim of this study is the development of macro policy instruments with a broad domain, applying to aggregates of many situations and many behavioral choices. In order to develop and assess such macro-environmental policies, models must preferably reflect that scale level. Some specific contributions to modelling this higher scale level will be made in Part Four. Here, their main set-up is worked out, instruments being defined in terms of these model specifications.

The general notion of modelling worked out here in Part Two, is the one Tinbergen (1967) applied in his famous book on macro-economic modelling, as a basis for macro-economic policy. Tinbergen differentiates the analysis of instruments at three levels:

◇ reforms, that change the foundations of social organization, "affecting the spiritual aspects of society and essential relations between individuals." (p.186)
◇ qualitative policies, that change the structure within given foundations, such as long-term policy change (p.149 ff.)
◇ quantitative policies, that regulate the position within a given structure, to be implemented frequently, at short notice (p.48 ff.).

This hierarchy in levels of generality may similarly be described in terms of the triad social order, social structure and social processes. In environmental policy, a substantial change in environmental liability rules would constitute a change in foundations; the introduction of an emission tax for a new substance would constitute a qualitative change; and the adjustment of tax levels and allowable emissions in permits are examples of quantitative policies. The three levels are related, obviously. Changing foundations opens up possibilities for qualitative changes in structures. Changing structures allows new quantitative policies. The main interest here is in reforms in foundations and in qualitative changes, the perspective being long-term policy development, that is strategic policy design. Any introduction of a new instrument either is a qualitative change or a change in foundations. Quantification is possible only in the contexts thus created. Some very preliminary quantification will be done at the case level only, in Part Five. In this part, the function of the main model is first to define instruments for environmental policy systematically. This is done by filling in certain crucial submodels i.e., those specifying the interfaces between governmental, societal and environmental processes. The strategic analysis on how the instruments will function in the full effect chain then requires general notions on main relations in these three submodels, especially those involving society and environment. At this general level of strategic analysis quantification is not yet possible. Quantified predictions on specific policies require a filling in of these submodels. The unknowns and disputable relations in the submodels of course cannot then be avoided. It seems impossible to develop these submodels a general level, for assessing any policy option. Quantified modelling will be case-specific. In the strategic policy design, here on instrument choice, such specific models can only play an illustrative role.

The next chapter first turns to the question of how the effect chain might be structured. Five main modules are distinguished, each of which may be filled in different ways. Two of them are required in defining instruments for environmental policy. They are the submodels that specify the government-society interface and the society-environment interface, see figure 2.2.1 on page eighteen below. How the other three modules might be filled in by related sub-models is indicated at a very general level only.
The following two chapters work out the two interfaces in more detail. The society-environment interface gives the analysis of environmental impacts produced by groups of processes. The government-society interface gives the prime working mechanisms with which governments may influence society. The combination of a mode of analysis with a prime working mechanism defines the main categories of both micro and macro instruments for environmental policy. The final chapter of this part on the general framework summarizes the conclusions of each of its chapters.

2.2 MODELS FOR POLICY INSTRUMENTS: SYSTEMS AND INTERFACES

2.2.1 Introduction

In modelling the effect chain of environmental policy, three related systems may be distinguished. First there is government that, as the main regulator of society, is also responsible for environmental policy. The government system generally consists of at least a political decision-making unit and a unit that takes care of implementation[1]. Internal organization may be simple, essentially consisting of one organization executing both functions. Or it may be very complex. There may be internal differentiations both within the decision-making unit and within the unit of implementation. There also may be internal rules fixing mutual rights and obligations of different governmental organizations. In Western societies, and most other ones, the latter is the case. Governments first make general decisions and then implement these in society in some separate process[2]. The purpose of implementation is to influence[3] society as desired.

Hence, secondly, there is society as the social system to be regulated by government. Within the social system, independent units, persons and organizations, are obliged to adjust their behaviour because of the policies implemented. To these social organizations also belong all those parts of government not directly involved in implementing the policy in question. The behavioral changes caused, through a single simple, or many complex processes, lead to changes in the way society influences the environment.
Thirdly then, there is the environment as influenced by society, itself another highly complex system. It is the resultant effects to the environment that is the aim of environmental policy. That prime aim is what turns policies into environmental policies.

For fundamental as well as practical reasons, a deliberate choice has been made here to analyze the sum-total of these three systems, that is 'the model', in a very limited manner only. One could analyze the interplay between the three systems over the course of time, in a dynamic analysis. For two systems, i.e. government and society, this is a quite familiar procedure. In the systems analysis of political life for example, see Easton (1965) as the founder of this type of analysis, there is a closed loop of causation, through feedback. Political decisions, through their implementation, influence society. The change in society leads to more, or less, support for the political system and to new demands on it, then to new decisions, etc. The closed loop is necessary for a dynamic, temporal analysis of politics. The loop connects two main systems only, government and society, as the environment, the third system here, is not usually considered an independent system in political science. In human ecology there is a similarly closed loop of two systems, see for example the environmental scientist De Groot (1992b). Society, as one

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1 Other subdivisions are possible of course, the most obvious one being that of Montesquieu. In decision-making, the legislative, the executive and the judiciary are involved; in implementation, the executive and the judiciary.

2 Decisions may be made fully by some separate decision-making unit or by the implementing unit itself, based on the aims set for that unit. In the former case implementing officials "go by the book" (Bardach and Kagan 1982); in the latter case officials have a greater amount of discretionary power.

3 I do not choose a particular vocabulary in the unceasing power-influence-authority debate. Using one specific vocabulary would lead to sophistic constructions, since part of power is to arrive at the binding decisions desired, another part is how power is exerted in changing the behaviour of individuals and organizations. Although the terminology is generally avoided, the subject cannot of course be avoided.

system, is linked to the environment, the second system, in a double way. It influences the environment by using it, and it is formed by the environment through the opportunities the environment presents or precludes. The opportunities as used by society in turn lead to a certain use of the environment. That, in turn, adds to or subtracts from environmental opportunities, etc. Luhmann (1989), a theoretical systems-oriented social scientist, similarly makes a main distinction between society and the environment as its ecological surroundings. The analysis of these two authors is also restricted to the mutual interactions of only two systems, in their dynamic relationships.

The fundamental reason to avoid the feedback loops and to opt for a linear causal string is that it best reflects the situation to be analyzed. The interest here is not the analysis of the political system, dynamically or otherwise. Nor is it primarily the developments in the environment. Nor is it the symbiotic relation between society and environment. Nor is it a dynamic, temporal triadic systems analysis, with government, society and environment reciprocally influencing each other. It is how the political system may, through its influence on society, influence the environment in the desired manner[1]. More precisely, the models developed should be suitable for comparing the instruments for environmental policy. It is the effects of adding one instrument, of replacing one instrument by another, and of changing existing instruments that are to be indicated, all other policies remaining equal.

The causal analysis strictly required for instrument and policy design thus goes one way only. There is one interface linking government to society, based on some characteristic of implementation. There is a second one linking society to environment, based on human interferences in the environment. Both interfaces are unidirectional. These interferences include resource extraction, emissions, and other disturbances[2]. This simplest causal string thus consists of three main systems, connected by two interfaces, see figure 2.2.1. Within each system, internal loops in causation cannot be avoided in the more complex models required for higher level instruments.

Some less than self-evident choices are implied. First, the environmental analysis covers environmental processes only; the causal chains stop there. Thus, the effects that health problems due to air pollution may have on labour productivity do not belong in the analysis. Secondly, the direct influence of government on the environment is a process in society like any other and is therefore disregarded in the analysis of the government system. Government thus influences the environment only through changes in society. Environmental problems due to social activities may be redressed by changing social processes; that is the only option in the model structure chosen. Direct intervention in the environment is not a separate subject calling for special attention. This reflects the subjective position taken here that such options should be considered only if prevention of

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1 A dynamic analysis of three independent systems would be extremely complicated in nature. Unquantified models cannot hope to cope with the conceptual difficulties arising. Instead of one feedback loop there would be three interconnected ones.

2 A conveniently short word to indicate all types of human influences on the environment is lacking. This state of affairs requires irksome formulations, like 'all inputs from and outputs to the environment', see hereafter. 'Impacts', an often-used word in this context, indicates both interferences and the further effects of these interferences in the environment. Discussion on this subject in the SETAC were of no avail. I cannot hope to introduce the widely-accepted new word here but I will give it a try: "*interferences*".

interferences cannot be effective enough. Thus, neutralizing acid rain through the spreading of calcium, fertilizing oceans to let them take up more carbon dioxide, and shielding the earth from the sun to redress global warming are only options of last resort. The main arguments for this position relate to the stability of the environment. If more environmental variables are fixed to redress the negative effects of human activities, the system will become less resistant to further disturbances. The end picture would then be the control room of the earth where the breakdown of one computer, or of one surveyor of the warning screens, leads to total disaster.

A practical reason for choosing a linear causal chain, without feedback loops, is one of simplicity. If the general system were analyzed as the interplay of three systems, a very complicated dynamic analysis would result. One autonomous change in society would lead to environmental changes that in turn would lead to political changes that would lead to further social changes, etc. A reduction in acid emissions may occur, for example through a shift to nuclear energy. Reduced corrosion would then allow a shift from aluminium to steel as an outdoor construction material, used quite widely in many outdoor applications. Thus coal mining would increase, as would all emissions involved in producing steel. These, in turn, would certainly induce other social and political changes, such as the increased use of the hydro power formerly used in aluminium production. Many public policies in the social and economic domain would be adjusted. Of course, all related environmental policies would be readjusted as well. Each of these changes in society and government would set in motion a new cycle of changes in all the subsystems of society. The only restriction on the exploding web of effects would be the length of time considered. The fading out of one initial change is not generally to be expected.

Thus four relations between the three main systems are being disregarded. They are the direct influence of government on the environment, and three feedback loops: from society to government; from environment to society; and from environment to government. The simple remaining causal string has a beginning, in government, passes through society, and has an end, in the environment.

In theoretical systems-oriented terminology, the general structure of the model is now extremely simple. First, it is a closed system, to retain some order in the extremely complex reality to be handled. Independence from its surroundings is how von Bertalanffy (1956), one key founder of systems theory, defines closed systems (see also Kramer and de Smit 1991, p.43). The closing might be argued on the grounds that the full environment is included as a separate element. This would make the empirical assumption that the system is isolated from its environment superfluous. Or the system may be analyzed as a closed one only to reduce complexity, conditional to the knowledge that more relations exist. The system thus simplified is to support decision-making on instruments and policies. In decision-making, the greater environment may be considered separately, in a more subjective manner. The second simplification is that there is no coupling, positive or negative, between the main elements as no feedback loops are considered between them. At this level, causation is thus modelled as a purely linear string, with one-directional interfaces.

The theoretical choice for this study on instruments is that instruments ***are*** *the interfaces of the simplified linear string model.* The working mechanism that sets in motion the changes in society, at interface I, is the first part of the instrument, the object and target of the behavioral change define the domain of the policy instrument. For environmental policy, the domain is then defined in terms of interface II, that between society and environment. This position might seem acceptable broadly. It may be more restricted however than usual. One major restriction is that instruments with an object that has no environmental content are simply not instruments for environmental policy. Any measure taken that influences society will also influence the environment. That is true for example for instruments in educational policy, macro-economic policy, and transport policy. If decisions on specific policies also reckon with the expected effects on the environment, these policies still remain what they are, according to the object and target of the instruments used. Economic policy, for example, may use the educational instrument to increase productivity and secure new markets or it may use capital depreciation facilities, for the same aim. Suppose that a choice is to be made, that the educational instrument is more beneficial for the environment, and that it is favoured, partly for that reason. That still does not transform extra outlays for education into an instrument of environmental policy. Of course, the society-environment interface may be defined broadly, including large numbers of societal processes in the interface. Many more instruments may then be seen as instruments for environmental policy. The broadness of this definition will lead to overlapping claims on instruments from other policy domains, such as that of economic and educational policy in the example. Also, following this approach will import the problems into instrument design of modelling substantial parts of society. Together these consequences of enlarging the interface would make a methodical scientific discussion on policy instruments, let alone a public one, virtually impossible.

The definition excludes some of the ecotaxes proposed now. Extra excises on petrol may be beneficial for the environment but are not an instrument of environmental policy. Lowering the value added tax on services might, or might not be beneficial for the environment. As there is no environmental specification in the instrument, it cannot be an instrument of environmental policy in the strict sense used here. However, an ecotax might also be specified in terms of the society-environment interface, such as the mixed tax on energy depletion and CO_2 emissions proposed by the European Commission. That ecotax then is a financial instrument for environmental policy. The definition of instruments for environmental policy thus splits ecotaxes into two groups. Instruments specified in terms of the society-environment interface are instruments of environmental policy, the others are not.

The definition also excludes some purely procedural instruments. Suppose that the obligatory environmental audit of the firm is restricted to checking that operations are according to permits and that the internal procedures of the firm are according to internal rules, e.g. that the environmental manager is heard duly before decisions are taken regarding the environment. Then that environmental audit is not an instrument for environmental policy in the sense used here. It would not even be part of the permit instrument. That type of audit could be a private reaction to the permit instrument, or to other instruments for environmental policy that regulate the firm. An obligatory environmental audit of this type could be seen as a government induced self-enforcement procedure, for other instruments. It would not be an independent instrument itself. Another example are the subsidies for environmental groups common in many Western countries. The environmental movement is not a unit at the society-environment interface.

Such a subsidy is thus not an instrument for environmental policy here. Failing to be an instrument for environmental policy does not imply that such policies are unimportant.

The next section begins with observations on the general nature of the relations in the several parts of the model. Then each of the three (sub-)systems and the two made-simple instrument interfaces will be described briefly.

FIGURE 2.2.1 THE THREE SYSTEMS AND THEIR INTERFACES IN THE MAIN EFFECT CHAIN OF ENVIRONMENTAL POLICY

	Interface I		*Interface II*	
GOVERNMENT	→	SOCIETY	→	ENVIRONMENT

2.2.2 Symbolic and material processes

The processes of thinking, learning, discussing, promising and deciding do not have any direct material effect, and therefore no direct effect on the environment either. They are symbolic in the sense that they manipulate symbols. They may function only to the extent that others recognize the meaning of these symbolic actions properly. Other processes, such as the production of food and materials, the use of energy, and the dumping of wastes, are material ones. Such material processes encompass a-biotic, chemico-physical, processes and biological processes, such as fermentation, digestion, respiration, etc. In human communities the symbolic processes determine or regulate the material ones, within the boundaries of course of what is possible in the material world.

Government, in terms of collective decisions and their formal implementation, consists mainly of non-material processes related to commitments, obligations, preferences, information, etc. Government, in its decision-making capacity is thus a symbolic type of system only. The environment here is a purely material system, encompassing both a-biotic and biotic processes. Of course, animals do process information and some may have intentions as well. For the level of analysis required for environmental policy design it does not make sense to distinguish systematically between the very limited symbolic aspects and the material aspects in the environment.

In private society, symbolic processes, related to commitments, obligations, etc., also exist. However, in society, there are also material processes, related to production, consumption and the handling of the wastes generated. So part of the functioning of society is related to symbolic processes, i.e., the symbolic part, and part of the functioning consists of material processes, i.e., the material part. The material aspects of government, ranging from housing to energy production and waste management, might similarly be treated as a material part of government. The consequence would be that both government and society would influence the environment directly. A choice has been made here to treat all material aspects of collective activities as part of society; not as 'material government'. Apart from simplicity, the main reason is that a clear and meaningful distinction between public and private economic processes is very difficult to define and does not make environmental sense. Is a private waste incinerator burning municipality collected waste at government set prices public or private? Is a coal mine with all shares publicly owned, and with a subsidy on its operations, public or private? Is publicly financed road building by a private company a public or a private material process?

So in society, in the midst of the causal chains between governmental, collective decision-making on the one hand and environmental effects on the other, a transformation takes place from paper rules, threats, promises, incomes, costs, incentives, etc. into the material effects of production, consumption and waste handling, both private and public. Only these material activities lead to diverse environmental effects such as climate change, desertification, health effects and depletion of resources. Thus, the effect chain may be split into a symbolic part and a material part, see figure 2.2.2. This does not mean that society consists of two independent systems, a symbolic one and a material one. The symbolic and the material form two aspects of one and the same society.

FIGURE 2.2.2 SYMBOLIC AND MATERIAL PROCESSES IN THE EFFECT CHAIN OF ENVIRONMENTAL POLICY INSTRUMENTS

GOVERNMENT → SOCIETY → ENVIRONMENT

← symbolic processes | **material processes →**

With the addition of this distinction in two types of processes, the position on the omission of feedback loops in the model may be stated more clearly. This omission of feedback loops towards government may be understood, first, in the literal sense that environmental processes, as physical entities, cannot influence governmental decision-making on instruments in any direct manner. Only the meaning attached to the effects can. However, there is no single right way to assess the meaning of environmental effects, nor is there a single right way to decide what the effects are. One problem to be solved is that empirical effects in the environment may cover a long period. Emissions of ozone-layer-depleting and climate-changing substances now may exert their detrimental effects in the atmosphere for several centuries. The consequent sea level rise also will be spread in time, lagging behind emissions in terms of decades of even centuries again. It would not make much sense to incorporate the feedbacks in a model based on real effects as expected in the course of time. These future effects can be predicted only by assuming future policies, in a society one can hardly know. The assessment of expected effects clearly is a complex matter, based at least on some notion of the importance attached to possible developments in the distant future. Similarly, the depletion of fossil fuels is probably a matter of many thousands of years, see the case study in Part Five. There is no simple rule on how depletion that might occur over such a long period of time should be translated into practical policies now. The mechanical feedback model where the state of the system is linked directly to specific corrective actions does not apply here.

Secondly, the lack of feedback loops in the model does not mean that in reality they do not exist. Environmental policy of course forms or creates such feedback loops. It may be based on the meaning attached to current environmental variables, such as the suffering of higher animals and the dying of forests because of acid rain. The main feedback loop, however, is that from expected or possible environmental effects that are to be prevented. The harm of ozone layer depletion is still limited but might become extensive if emissions go on. Climate change because of human emissions is not yet taking place, at least not provable in a strict scientific sense. It might already be occurring, however, in a hardly reversible manner. Policies that take into account such effects are implemented now, rightly I think. They form part of a culturally determined feedback loop. Such feedback loops towards government are left out of the analysis here because it concerns how rational governments should act now, taking into account all these existing, expected, and possible effects.

The instrument discussion is thus a discussion on how the feedback loop should be constructed, as a design process. It is not a question of the empirical nature of the feedback loop. The more modest aim of this study is to see how a rational feedback mechanism may be built now, for current and expected environmental effects, in terms of the instruments to be used. However, if preferred, the predicted results of the environmental analysis of applied instruments and policies may also be seen as a first step in the analysis of the empirical feedback towards society and government. The interesting subject of the strategic design of our long-term relations with the natural and physical environment is simply not the prime subject of this study.

2.2.3 Models of government

Models of government may include the full political process, from agenda formation to decision-making, implementation, monitoring and control, and enforcement. For the present analysis such a model would be too encompassing. Including agenda formation would implicitly include at least part of the feedback loops from environment to society and government. The function of modelling government here is to help define and evaluate instruments for environmental policies, that is to support choices on policies *within* the political process itself. As indicated, modelling how decisions on environmental policy factually evolve is not the concern here. The model of government relevant is one that supports that process by indicating the effects and normative meaning of possible policy decisions. Thus, interest theories of political decision-making currently popular in political science and economics are not relevant to the analysis here. Essentially, the questions to be answered are those someone would ask who is choosing instruments for environmental policy in as rational a manner as possible[1]. That person would want to know the main set-up of implementation, as a model of how political decisions might be effectuated by government. Implementation might be looked upon as a broader and also social process that may be quite independent from the central decision-making part of government. In that perspective, networks of power and influence, with each actors playing his part according to his script, may be analyzed for good reasons, both within the bureaucracy and in its relations to society[2]. Such an analysis may give insight into the general possibilities and limitations of bureaucratic implementation. For the purposes of

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[1] This does not exclude the possibility that environmental policies are used to further other interests, e.g. industrial interests. If environmental reasoning does not first have its autonomous development such actions would constitute industrial policy only.

[2] See for example Glasbergen 1989.

specific policy design and decision-making, and even more so for strategic policy design in terms of instruments, this type of analysis is far too complex. The simplest type of model of government, that also satisfies the needs of this study, is based on one supposition. I assume that decisions on main instruments and main policies are implemented the way they are meant to be, that somehow the primacy of law and government over the bureaucracy is upheld. It may be assessed of course if specific instruments are more difficult to implement than others.

This practical stance also avoids a danger attached to detailing the processes of implementation excessively. If used for policy design, detailed models have a strong bias towards regulation in terms of the details of the concrete bureaucracies in concrete situations. Also, it would seem quite impossible to regulate in detail the environmental aspects of each and every concrete situation. The current study has been set up to avoid that dead end of policy development. Policy instruments of a generic, macro type cannot be analyzed in such detailed networks in a practical way[1]. Implementation will therefore be treated here in a formal manner, at a very general level only. It is the main part of governmental processes, as a public activity preparing actions towards the mainly private part of society. Some observations on bureaucratic implementation will be made in section 2.3.4 on policy instruments. Only a very limited part of implementation will be incorporated into the interface definitions of the instruments.

2.2.4 Models of society

Society as a very general entity has to be structured for the sake of the analysis at hand. The now dominant loosely functionalist approach will be used to its purpose[2]. In the functionalist approach, any stable social system has functionally differentiated institutions for collective decision-making and goal attainment, here called government. For reasons of policy instrument analysis, that part of government engaged in environmental policy is treated here independently from society, not as part of it. Through collective decision-making, but also through internal 'autonomous' developments, the structure of society forms and changes[3]. The structure is the sum total of all stable positions and their relations, as fixed in rules and institutions, see below for a fuller explanation. The family, with the duties and obligations defined, is a main institution in the social structure of a society. Knowledge, values and ideas, are the major cultural elements of society. Partly autonomously and partly through collective decision-making, society deals with their creation and diffusion, in various specialized institutions (e.g. schools and universities) and non-specialized ones (e.g. families and firms). The sum total of all knowledge, values and ideas forms the culture of society. All activities in society that are related directly to its material support form the economy. The economy incorporates elements of both

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[1] See my comments on Glasbergen in Huppes 1990.

[2] Parsons 1951, as one very influential functionalist social scientist, has been 'translated' into a somewhat easier terminology by Johnson 1961. The latter names the four main functional requirements for social systems that remain existing over time: 1. Pattern maintenance and tension management; 2. Adaptation; 3. Goal attainment; 4. Integration. The emphasis on factors contributing to stability has given the functionalist approach a conservative appeal. In the sixties, however, action theory directed at change, with Etzioni as a main proponent, used the same analytic structure to formulate an action programme for social change, quite in line with Parsons' original intentions. See Etzioni 1968. The terminology of the functionalist approach is now used, more or less loosely, by most social scientists.

[3] The terminology diverges somewhat from Parsons' action system terms. 'Structure' there names the social system, with culture and social system as two main independent elements.

structure and culture. The decisions associated with these economic activities again are partly collective, related to government activities, and partly private, autonomously in society.

For environmental policy analysis the precise meaning of 'economy' is of central importance, as economic activities are the main cause of environmental degradation. In all modern societies the material requirements for the functioning of society are served by certain specialized institutions, but also by less specialized ones. Economic institutions specialize in the procurement of goods and services, i.e. production, the first main economic activity. Firms, as relatively independent economic organizations, produce most goods and services in industrialized countries. Whether they are owned privately or publicly makes no difference here for the analysis of their environmental effects. The ownership structure of economies of course is extremely important for their practical functioning, as may be seen in the case of Eastern Europe. Households also have a productive function, in industrialized countries mostly a limited one. Consumption, the other main economic activity, however, is generally quite diffuse in households, and in some private and public organizations. In their capacity as producers and consumers, households belong to the economy of society, even if not functionally differentiated and specialized. Waste handling, including reuse and recycling, and final disposal, as complements to production and consumption, are also economic activities.

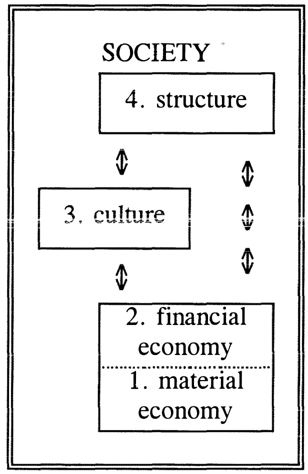
For assessing the effects of instruments and policies a basic distinction is made here between the material aspects of the economy, that are connected to environmental variables, and its "steering" aspects, at the symbolic level, that are connected to policy instruments. Economics, as a discipline, deals almost exclusively with these non-material steering aspects of the economy, as part of the symbolic aspects of society. This is true even if some often-used terms seem to indicate otherwise. Terms such as national product and private consumption refer to monetary aggregates, and not to physical entities. The price-level adjusted gross national product is an index with many worthwhile applications, such as in the analysis of aggregate unemployment. It does not say very much about material variables however. It is at a micro level only that prices relate to more concrete entities, such as specific goods and services. Economics also has branches that deal with non-financial aspects. Institutional economics takes into account broader aspects of societal decision-making, at the boundary between structure and economy. It concerns possible mechanisms for coordinating economic activities, such as several forms of markets, but also the relation between markets on the one hand and the hierarchical command-and-control mechanisms that characterize many decisions in larger firms, on the other. Power and influence are the means of coordination of economic activities in the latter case. Public, collective activities tend to be regulated more by such non-financial means of coordination. They are important in larger private firms as well. Such non-financial aspects also are symbolic.

The economy has a material basis of course, which is the characteristic that distinguishes it from the culture and the structure of society. Economics, as a discipline, hardly deal with such physical aspects at all. The sciences that now deals with the material aspects of the economy are primarily the applied physical sciences, such as chemical technology, biotechnology, metallurgy and agronomy. They generally consider only physical aspects related to specific functions. These material aspects of the economy are not usually relevant in economics for their biotic and a-biotic chemico-physical (including spatial) aspects. It is only these latter aspects, however, that are relevant environmentally. It is the functions of the physical entities, only partly related to these physical aspects, that count in economics and are the basis for monetary value. From which clays a glazed kitchen floor tile has been produced is not a concern in economics as long as the tiles 'function', that is provide their service properly. At the purely physical level the choice of clay may be highly important environmentally. Thus, functions or services form the main link between the financial part of the economy and the entities at the physical level that are relevant for environmental analysis. Economics, as a discipline does not fill the gap. Simple products defined in terms of material composition only, such as sugar, might be specified in terms of physical entities quite easily. Complex and technically variable products, such as cars or computers, can hardly be described in their purely physical aspects but only in terms of the functions they fulfil and in terms of the value they possess based on these functions. At the micro-economic level purely physical aspects may sometimes play a dominant role, but only indirectly. The physical properties of cadmium and zinc have led to their combined occurrence in the earth's crust. The supply of cadmium is inelastic mainly because of this combined occurrence, in low concentrations, with zinc. The connection between the financial and functional aspect and the material aspect is there, in every product and installation. At a scientific level this connection is still very weak. It is a challenge both to environmental scientists, physical scientists and economists to fill this gap. Filling the gap to some extent is a prerequisite for the design of instruments for environmental policy and therefore is a main task here.

In the domain of the economy one thus can distinguish between two levels. One is a purely physical or material level, to be described in terms of the physical sciences. The other is a symbolic, mainly financial level, to be described in the financial and functional terms of economics. The economy is divided accordingly into a *material economy* and a *financial economy*. Expressed in brief, a main analytic breakdown of social processes will be made dividing them into four main types related to the following: the material economy (1); the financial economy (2); the culture (3); and the structure (4) of society, see figure 2.2.3.

1. Material economy
The material economy forms the material basis of society. For society it deals with processes ranging from the extraction of resources, production and consumption, to all related waste handling and the disposal of final waste. Included in the material economy are its relations to the environment in terms of the inputs required from the environment and the outputs supplied to it. The analysis of flows and stocks of substances and energy is the most basic subject. The sub-discipline of material flow analysis is now developing rapidly, see v.d. Voet et al. (1992) for a survey based on a primarily environmental perspective. The IIASA publication of Ayres et al. (1989) gives the state of the art from a rather economic point of view. The challenge of the emerging discipline of materials or substance flow analysis is to connect this material basis of the economy to sensible units in the environment on the one hand, and to sensible units of the financial economy on the other.

FIGURE 2.2.3 FOUR MAIN ELEMENTS FOR MODELS OF SOCIETY



2. Financial economy
The symbolic economy is the mainly functionally differentiated part of society that directly regulates, or determines, the material production and consumption of goods and services. Central to the symbolic economy is money, the financial unit for expressing values, rights and obligations and as the basis of markets. The financial economy consists of producers and consumers in the form of decision-making units, markets and other institutions that regulate the behaviour of decision-making units, and functions and values of products, i.e. of the goods and services that the economy is all about. Governmental organizations with regulating functions of material production similar to that of firms also belong to the symbolic economy, as do public decisions on collective production and consumption. Examples of the latter are decisions on transport infrastructure, public transport, water cleaning installations, and recreational facilities. The role of money is so dominant even there that, also for that reason, such collective aspects are included in the financial economy.

3. Culture
Culture is the totality of knowledge, values, norms and beliefs that direct the behaviour of decision-making units. These units may be individuals, groups or even the totality of society. Education, exchange of information, and research keep a culture dynamically alive. One part of culture most directly related to environmental problems and policies is developing primarily in research institutes. Firms, governmental departments and publishers of papers and books are institutions that support the development of the culture of society as well, as do discussions between members of a household. Culture is the societal element most difficult to grasp and to control. Its role is broad. All demand for products exceeding the purely physical level of keeping alive is dominated by cultural factors. Changes in technology as applied physically, originate at the cultural level, through research and development. Conversely, all wilful changes in the structure of society are also based on cultural beliefs and values.

4. Structure[1]

Social structure encompasses the more stable patterns of relatively independent roles and institutions, and the stable relations between these roles and institutions. The functioning of institutions is guided by norms or rules for the behaviour of individuals and organizations. These norms and rules form part of the structure of society. The rules may be formalized, as in public law (tax laws, penal codes, court procedures) and in private law (property rights, liability rules, contract law). Organizational structure refers to the stable relations between organizations as functionally differentiated institutions. The structure of society may be described in aggregate terms, as macro elements, or in more specific terms, as micro elements. High level macro analysis of the structure of society relates to such topics as the organization of interest groups, the relative importance of different types of taxes, the predominance of the public sector in the economy, etc. Relevant to the environmental behaviour of firms at this level are the restrictions on liability limiting the protection of collective property rights on environmental goods. At a more micro level environmentally relevant elements of structure are the status of the environmental manager in some multinational, the content of the environmental audit as is emerging now, and the restrictions on international trade as in chemical wastes.

The general social structure is connected in many ways to the functioning of the economy, to the factual possibilities of environmental policies and to their outcomes. In China, for example, main elements in the structure are similar to those in Western countries. Production takes place in firms, and local and regional government agencies implement environmental rules. Non-compliance with these rules may lead to substantial fines there. However, in many situations it is precisely these local or regional governments that own the firms concerned. The distinctive structural element is that government then effectively may run such a firm, while also implementing the environmental rules. In that situation, the fine effectively is not paid by the firm but by the government itself, as a lower profit or an extra loss of the firm. Thus government is collecting exactly as many fines as it has to pay. It only makes payments to itself. The

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[1] Structure is a term with many meanings, see Pen's 1974 survey with at least seven different ones, see the following list:

1. Structure as relation, with isolation as the negation or inverse;
2. Structure as observable regularity or pattern, with random data as the inverse;
3. Structure as order from a normative point of view; with chaos as the inverse;
4. Structure as capital, both encompassing physical capital, human capital, and the collective variant of the latter as institutions. Pen states function and process as possible inverses (p26), I would prefer lack of capital.
5. Structure as the constant or unchanging, with the variable as the inverse;
6. Structure as the possible or potential, with the actual or factual as the inverse;
7. Structure as the total of all causes, with partial relation in a substrure as the inverse.

Following Parsons, interpreting him quite differently from Pen, structure as defined here would be placed in number 4, the capital variant. Parsons (1951), in his chapter on the theory of social change, quite explicitly states that the structure, although less variable than the processes functioning in it, is not a constant, and cannot even be one. For many lower level, short term types of analysis, however, it may quite sensibly be treated as a constant. 'The model of society' does <u>not</u> imply structure in the number 5 sense usual in quantitative modelling, as the given parameters that define the quantitative relations between variables.

fine, as a sanction on non-compliance of behavioral rules, thus cannot be effective in such a structure, nor can the emission tax[1].

How are the four different elements of the model of society related to one another and to the interfaces? The boundary of the material economy to the environment is worked out below, as the society-environment interface. It is at the boundary between the material and the financial economy that the symbolic social processes and all the material processes of society meet. A steel mill produces an amount of plated steel with a certain tensile strength, resistance to corrosion and hardness. That makes it fit for certain higher level functions, such as in the body of cars. Based on such functions it has a certain value in monetary terms. The values, and partly also the functions, form the symbolic part of the process. The composition of the steel, as an alloy, with a certain type of bond between the iron and the other elements, form an aspect of the physical reality, and as such create the tensile strength and other features of the product that are necessary for its functions. Thus, the material aspect forms the basis for funtions, with the functions in turn forming the basis for the value of a product. Each material process in the economy may be described in these three different ways. It transforms values into values, as cost into proceeds; it transforms functions into other functions, as heat into power; and it transforms physical entities into other physical entities, like hydrocarbons into CO_2 and water. For all material economic processes there is a direct link with the symbolic economic aspect in terms of values, with the functions forming the link between these two. For cultural and structural elements in society there is no such direct link. They may change without being accompanied by directly corresponding changes in physical reality. They form systems of their own, having an indirect bearing on the material economy only.

Next to be indicated are the boundaries of the financial economy with the symbolic aspects of society, in terms of culture and structure. First, the financial economy is clearly that part of society where financial aspects dominate as indicators of values and are related to functions and through them, to material flows. It also is clear that in the mainly monetized modern economies nearly all dealings with material entities are influenced heavily by this financial aspect. A first question is whether non-financial aspects related to decisions in production and consumption should be included in this financial economy as well. The position taken is that they should, but only partly. Decisions on production and consumption belong to the economy. Ideals on how to spend holidays, although influencing economic choices and creating specific demands in the economy, do not. They belong to the cultural domain. Research on technologies is a borderline case. General, not yet product-specific research would belong to the cultural domain; the applied research for specific applications, to the financial economic domain. Boundaries are not sharp here. Market structure, e.g. an oligopolistic system versus a monopolistic system, as in the case of electricity production in different Western countries, would belong to the structural aspect of society in so far as this structure is

[1] See Vermeer 1990, pp 22-24, on fines in environmental policy in China. If the fine or emission tax is paid out by a different body from the one that owns the firm, the problem is solved, at least if these public bodies are independent of each other financially. Fines tend to be used as emission taxes because of their proceeds to governments. Recently, China announced the introduction of a substantial emission tax on SO_2, according to a report in China Daily, as cited by the Beijing correspondent of the Dutch daily NRC-H, 19-2-1993.

created by judicial rules by institutional relations between banks and producers of capital goods, and by the organization of larger energy users. Given all these institutional factors that are, or create the structure, the functioning of the oligopolistic market, as a group of related decisions on amounts of goods and services and their prices, is part of the financial economy.

How can this general outline of the model of society be used practically? First, it can show the levels of models of society that may be used in analyzing the environmental effects of instruments and policies. The simplest type of analysis almost restricts itself to the level of the material economy, but not completely. The small part of the model of the financial economy added is a very simple one. It assumes that somehow the physical reality will be changed according to the governmental decision as stated, as in permits. It abstracts from nearly all social processes of a financial or broader symbolic nature. The minimum symbolic step required is that firms will follow the orders given to them, e.g. they will behave according to the limitations stipulated in permits. One first step towards complexity is the incorporation of the financial parts of the economy more fully, the introduction of prices, markets, functions, etc. Models thus become much more realistic, but also more complex. If some environmental measure raises the costs of one product, the quantity produced will drop and that of several other products will increase. A next step is to incorporate cultural elements into models as well. People like to show how much they care about the environment. Ecolabels on products, indicating the relatively minor effects of some products on the environment, can thus be part of the cultural steering mechanisms that policies may activate. The additional inclusion of structural elements may create the most "realistic" models, as in the incorporation of effects on the size distribution of firms and the related market structure.

Two examples will be sketched to clarify this point further. First, how may the effects be modelled of making obligatory a desulfurization apparatus of a coal-fired electricity plant? The model should at least specify its operation at the material economic level. It is assumed for example, that the apparatus will effectively be installed and will work according to specifications. Neither of these assumptions is well-founded. The model may also include broader financial economic variables, e.g indicating possibilities for lowering costs by behavioral adjustments like circumventing installation or the costs of operations, but also indicating shifts to other such production techniques as wind power. Such options become more attractive because the price of coal-based electricity now reflects the costs of desulfurization. At the cultural level, a spurt in research on solar-based production technologies may next be expected because these may become competitive with the now higher costs of fossil energy. Finally, structural elements may be added in the analysis of policies. In the causal sequence from electricity plant desulfurization to wind power, and to new research on alternative energy technologies, a resultant structural effect might be the breakdown of the current monopolies in electricity production, through pressure from the interest groups representing these other sources of electricity. The broadening of the model here leads to ever-increasing environmental improvements. In this example, the causal chain started at the obligatory introduction of flue gas cleaning apparatus, at the material level. A causal chain may start anywhere. The forced breakdown of monopolies in energy production is being introduced in several Western countries now, though not primarily for environmental reasons. When starting at this structural level, the model cannot avoid structural variables. Achieving environmental effects, however, at least requires model mechanisms stating the workings of the financial economy and, through this, the processes involved in the material economy. The causal lines in society sketched thus far are quite simple strings. Going back and forth between different types of societal processes may transform these simple strings into webs of increasing complexity. In a next round of effects, dynamic increases in efficiency will quite probably first result in diminishing emissions even further, but next, through the severe drops in prices induced, may lead to a sharp increase in all electricity-consumption related activities. There is a nasty choice to be made between complexity and increasingly conditional truths, and clear and simple but perhaps false results when modelling the effects of instruments and policies.

The second example concerns the Californian requirement that by the year 2000 ten percent of all cars sold should be emission free[1]. This policy measure may be analyzed at these several levels again. At the nearly physical level, 'ten percent of all cars sold are emission free in the year 2000'. An estimate of sales tells the effects on car emissions, as compared to the same number of all cars sold with combustion engines. Emissions will decrease by ten percent in this model. Modelling effects in their larger economic setting, through the related financial parts of the economy, may picture a much more limited effect. Emissions at other places in the economy will increase because of the extra electricity production required, because of the production and disposal of extra batteries (e.g. lead, cadmium, lithium), etc. There is a serious possibility that this analysis will show a negative effect of this policy measure for several types of environmental impacts. It is even seriously possible that overall environmental effects will be negative. The next model elements however, the cultural and the structural ones, together might reverse the net negative outcomes of the economic analysis. The volume of research and development on the decentralized use of secondary energy, for example, may increase dramatically, giving a boost to the possibilities for solar power. Social organization may change, to allow the effective introduction of new systems of energy use. These combined financial-economic, structural and cultural mechanisms come closest to the full, but mostly unknown, reality. They are therefore the most difficult to model, both conceptually and quantitatively. They may show the broadest effects on the material economy and through it, on the environment. The choice of level of analysis is crucial in assessing the pros and cons of this or any other policy measure.

The complexity of models may be increased further by including the non-environmental parts of politics as well. Macro-economic consequences may or may not be modelled independently of these broader socio-political processes. The Hermes model used by the European Commission for assessing the effects of policies does not explicitly assume effects on political decision-making. By contrast, the model of the Dutch Central Planning Bureau used by the Wolfson Commission (CPB 1992), see appendix 5.5.2.1, does incorporate some feedback through the non-environmental parts of government. In it, energy taxes lead to price increases that multiply since the social partners, including government, will not be able to restrain themselves in trying to shift the increase in the tax burden to others. In the Netherlands, formalized tripartite negotiations are held yearly

[1] Production of a car takes a substantial amount of total energy use in the life time, as one indication of environmental effects. Moll 1993 gives data that indicate a level of over 10 percent.

between organized labour, organized business and central government on price levels, income increases and related tax adjustments. In the CPB model, these negotiations are expected to collapse because of the severe price changes resulting from the abrupt introduction of a high level energy tax. Such feedback loops to government make it difficult to assess the effects of one particular environmental policy measure. Models of this structure may show the combined effects of several policy measures together, as a broader policy scenario. They then are not suitable for instrument analysis.

Ideal, fully macro models of society would state the environmental effects of any policy measure in terms of its effects on all environmental interferences, grouped for example, according to the environmental problems they are related to. All four main levels of variables would figure as intervening variables. Such models are completely lacking. Quantified models that are available only cover parts of the financial economy and the material economy. The macro models available, all financial economic ones, lack the technological specificity to show material changes. At best they specify the material aspect of production and consumption through sectoral emission factors, see Hafkamp (1991) for a survey of such models. On the other end of the spectrum of economic models, there are micro models for environmentally relevant goods, such as oil and cadmium batteries. At best, such a micro model is connected to some macro model. A primary example is the macro-economic energy model of the Dutch Central Planning Bureau used for the analysis of energy taxes. It is discussed in Part Five, in the case on energy and global warming. Micro models of the financial economy, relating to one activity, one product or a group of products, are more widely available. Models of the material economy, e.g. substance flow analysis, see below, are being developed currently. At best they work at a meso level, covering all flows of one group of substances through society.

Complex, multi-level models of society will be disputed fiercely both because of their more speculative nature and because of the interests connected to their outcomes. There is, however, a quite large body of broadly accepted knowledge on these more general aspects of society. Certain characteristics of the structure of Western society will be generally accepted. The social order is mixed. Markets primarily decide on the allocation of productive inputs. Private firms decide on technologies and products. Consumer preferences play some independent role. Markets are mainly competitive, to some extent. Government mends what it sees as main defects in the functioning of markets, or at least tries to do so. It regulates aggregate demand; dismantles monopolies, nationalizes or stringently regulates natural monopolies such as utilities; it is active in the labour market and in technological development.; and it has a main stake in education. There is a substantial redistribution of income, etc. Although heavily influenced by government, however, the structure, culture, and economy of society also develop quite independently of these active governmental interferences. Such general notions can hardly be incorporated in quaitified, formalised models. Cultural relations seem relatively underdeveloped. There are some broadly accepted notions on the interest-related nature of behaviour and on the role of ethics, but also on the slow changes possible in standardized behaviour. At the other end of the spectrum, formalised micro-economic and macro-economic analysis are widely available for modelling society in its only financial-economic aspects.

Modelling society may thus be done in an extremely precise but simple manner, restricted almost entirely to the material and financial level. This disregards the symbolic processes that lead to potentially much more important indirect and secondary effects in the long term. Or it may be done in a complex manner, taking into account the interplay between material economic factors, financial economic factors, and cultural and structural factors. At that level relations might also be quantified but this quantification can hardly be based on more than sketchy, qualitative knowledge. There are of course, all shades of complexity in between. It is sufficient for the present discourse that models are distinguished according to the type of relations they consider, at the four levels specified, and according to a micro-macro dimension, with the higher level models potentially being more macro. Four main groups of models may be singled out according to the four types of factors distinguished: those with scarcely any financial economic relations at all, those that at least specify some relations in the financial economy, those that also include cultural relations, and those that add effects of structural changes in society. Cultural and structural variables never influence the material economy directly. It is only through the financial economy that they can exert their influence at the physical level. The physical level itself is to be modelled as well, encompassing at least the processes whose changes are induced by changes at the financial level. The material economy has its own types of empirical relations, based on natural science. Matter and energy, for example, cannot disappear but can only be transformed. Input-output models form the simplest macro models for both the financial and the material economy. The economic input-output models, pioneered by Leontief, are widely used, for environmental modelling as well. The analogue of the material economy also works with fixed input and output coefficients, the substance flow analysis (SFA) at the societal level. It does not, however, state these coefficients in monetary terms but in physical terms, for one substance or for a group of similar substances. Another difference is that the units used have not been formalized as sectors but as groups of similar processes that differ according to the substance investigated. This type of model will be treated in some detail in Part Four.

If the main line of argument is again considered, the question arises of how these different levels of modelling relate to the macro character of instruments. A simple ordering can be carried out. Instruments that directly induce structural changes in society by their nature may be the most macro in character. Structural models are required for the analysis of such macro instruments. Cultural instruments may be nearly as broad, or macro, but might be restricted to a more limited, less macro, domain as well. Thus, models incorporating cultural but not structural variables may be somewhat more restricted. Financial instruments require at least a model covering financial relations. Economic models incorporating only such financial relations may be more restricted still, but need not be. The simplest instruments regulating only specific processes and products, which are related to models with almost no symbolic relations, can hardly be macro in character. The level itself, however, does not fully fix the position on the micro-macro dimension. Although the higher level instruments and model allow the nature of instruments and models to be increasingly macro, a micro approach is perfectly feasible at these levels as well. Liability for damages to marine life because of oil extraction in in coastal waters is a US example.

After this sketch of the options and impossibilities of models for society, a discussion first follows on the implementing aspects of instruments, at the government-society interface, before turning to models of the environment. Then, after some broad picture of environmental models in section 2.2.6, I will treat the specification of the society-environment interface, the second element in the definition of instruments for environmental policy. Only the addition of this latter element gives policy instruments a content, transforming general instruments into instruments for environmental policy.

2.2.5 The government-society interface: policy instruments 1

Environmental policies must influence society to become ultimately effective in the environment. Governmental processes must somehow be related to the processes in society. Policy instruments are defined here as forming the government-society interface. Thus, each instrument has its own interface specifications. The main task in modelling this interface is to show how the relations may be shaped, which choices are possible and may be attractive. The interface does not necessarily make up a separate reality. It only describes the way the connection is made between the two empirical systems. Since bureaucracies involved in governmental implementation may be adapted to instrument requirements more easily than private society, it would seem most practical to model the interface starting at the society end. The government part can then be added in the detail required. The types of social mechanisms present in models of society may also reflect the types of instruments that may be used to influence society. The four main types of processes as modelled above may indicate four main types of instruments. Other societal and governmental aspects may then be used to make further sub-classifications of instruments. The main criterion for the classification of instruments is thus the direct, or prime working mechanism an instrument may have in society. It is not their judicial form, for example, law versus discretionary decisions of officials. Nor is it the main object of policies, as in product policies and process policies. Nor is it the main subjects of policies, as in the different policy target groups that may be distinguished. Nor is it their main working mechanism. That may depend entirely on the type of model used and how it is quantified.

1. Prohibitions

Behavioral rules stated primarily in material terms, line one in figure 2.2.4, define the *prohibiting instruments*. Several other names are used, such as physical instruments for environmental regulation, direct regulations, and regulation by law. Physical instruments have the connotation that the instruments themselves are physical, such as the fencing off from the public of areas not to be walked into. Such physical instruments are not meant here[1]. It is not the instruments themselves that are physical. It is what they prescribe or proscribe that is stated in physical terms. Financial instruments might specify the same physical entities however but tax or subsidize them. The term physical instruments will not be used for these reasons. Direct environmental regulation is ambiguous as to what is direct: the description of the desired behaviour, the environmental effect, or the application of the instrument, to name a few possibilities. Therefore, this term will also

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[1] Some instruments could be defined at that physical level, like fencing. Blocking the entry to a nature area with barbed wire is an example. However, such nature oriented policies are excluded here as a subject. Supplying disposal facilities for chemical waste is an example not excluded as a subject. The regulating mechanism there may be the prohibition of other types of disposal of such waste. It too may be purely cultural. Still, it might be attractive in some situations to include the physical act itself as an instrument. Such instruments then would require further differentiations in this general conceptual framework. They are not worked out here as they could hardly be macro instruments.

be avoided. 'Regulation by law' and 'juridical instruments' are most unapt terms. Structural changes, such as liability rules, may be based on law. Taxes are nearly always based on law. On the other hand, prohibitions may be enacted without laws, through private contracts, as happens in the Netherlands in 'covenants'. Primary examples of the behavioral rules concerned are general prohibitions on products and individual permits for the functioning of processes. The simplest model for this type of instrument assumes that nothing will be done that is not allowed. Thus the symbolic social process that initiates effectuation is *prohibition*. Since it is the type of symbolic mechanism, as the prime working mechanism, that should give its name to the instrument, the term 'prohibiting instruments' or prohibitions[1] is preferred here to alternatives such as physical regulation, direct regulation, command-and-control techniques, etc.

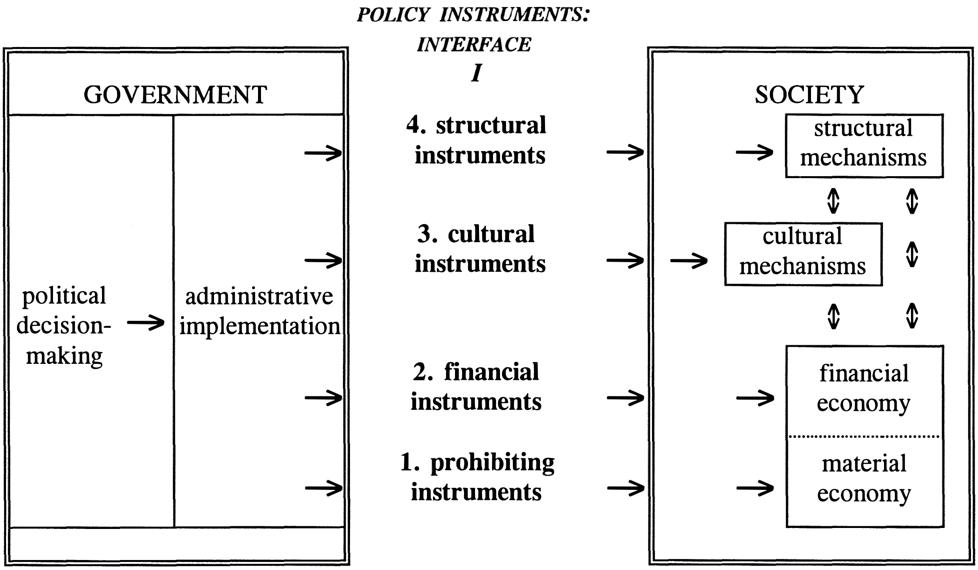
To avoid a common misconception, it should be pointed out that prohibitions are not clearer in their effects then other types of instruments. That is the case only seemingly because clear but unjustifiably simple models can be applied to them. Prohibiting instruments may be analyzed in such a narrow way, assuming that the only change in society is the one stated in the instrument, with all or most of the subjects regulated acting according to the rules as stated. The instrument then presumably covers all relevant processes in society. Apart from its simplicity and ease, there is no good reason to use such an unrealistic model. In the first place, prohibitions do not work all that simply. Some rules are followed and others are not. Moreover, broader economic effects can never be avoided. If one process has to add some cleaning apparatus to its flue gases, other processes and products will become more attractive, leading to similar or other environmental problems. Cultural changes are always inevitable, as in the case of the altering attractiveness of technological developments and a shift of attention of managers to stated rules instead of aims, for example. If only marginally, prohibitions always contribute to structural changes, for example by lowering the share of the sciences in higher education through the less attractive nature of bureaucratized science jobs. For a comparison to the other types of more macro instruments, more complex models are required, incorporating one or more of the financial, cultural or structural types of mechanism. Modelling these effects is extremely difficult. Some rule-of-thumb modelling might indicate whether such indirect and secondary effects can become dominant.

2. Financial instruments

Instruments that directly influence the main symbolic economic variable, prices, are *financial instruments for environmental regulation*, type two in figure 2.2.4. Financial instruments are restricted here to those that directly change costs or prices, such as taxes and subsidies. Another term could be economic instruments. For reasons of analogy to cultural and structural instruments, the term financial instruments is slightly preferred. Financial instruments for environmental policy cover taxes on emissions and subsidies on emission reductions. Many different financial instruments may be defined, however: government subsidies on the acquisition of nature reserves by nature conservation organizations, tax deductions for donations to green lobbying groups, payments for having

[1] Pen (1971) gives a taxonomy of methods for anti-pollution policy, with prohibition as number one. However, several of the other six are based on prohibitions in the broader sense used here and would fall here under the same general heading.

FIGURE 2.2.4 THE FOUR MAIN TYPES OF POLICY INSTRUMENTS, BASED ON THEIR PRIME EFFECT MECHANISM IN SOCIETY.



rare birds nesting or rare plants growing on one's property, withholding agricultural subsidies from plant types that require many pesticides; etc.

Financial instruments and economic instruments are fully equivalent terms here. Other instruments might be seen as financial or economic as well but are classified otherwise here. Which boundary cases are excluded? Some policy measures may employ price changes as a major working mechanism, especially through restricting marketable amounts. Fish quotas for fishing at sea is an example. Such quotas are seen here as prohibitions, since the prime working mechanism is to forbid catching fish. Do they become financial instruments if trade in quotas is allowed? Even then, the tradable fish quotas or tradable emission permits are *not* seen as an financial instrument of environmental policy. It is the general prohibition to fish or emit that is the primary working mechanism, with the secondary exception that allows fishing or emitting to certain persons only, e.g. those that bought a permit. Another reason to call tradable permits a prohibiting (and not a financial) instrument is that trading may be seen as one method amongst others to find the least cost compliers with prohibitions. Trading is thus economic in the sense of 'efficient', it is not an economic instrument in its primary working mechanism at implementation.

Auctioned emission permits or fishing rights are not yet seen here as having a primary financial working mechanism either, but they come nearer as allocation is influenced more directly by this mechanism. Firms that cannot buy at the auction will have to shut down. The difference with financial instruments becomes even smaller if the auction is for permits of restricted duration and a minimum price. If, as a next step, as many fixed-priced permits are sold as the market will bear for that price, in small units, available at any time, then the instrument has become a financial one. The fish quotas then have been transformed into depletion taxes and the emission permits have been transformed into emission taxes. If the last step is not taken, the prime working mechanism is still a prohibition, and the main working mechanism an economic one, as is true of most other non-financial instruments. This does not mean that adding such trade or auction aspects to quotas or permits is not sensible. On the contrary, using markets in this way may help the issuing bodies arrive at a minimum cost distribution of quotas and permits among the firms concerned.

With financial instruments and with other types of instruments as well, the functioning of markets is a central modelling feature for predicting the effects of applying such instruments. Financial instruments may work more broadly, at a more macro level, than prohibitions. General emission taxes on a certain type of emission may encompass a range of activities that cannot be covered by a prohibition. Of course, the range of financial instruments may also be very limited, as is a grant to one firm for some especially important investment in environmental technology.

3. Cultural instruments
Instruments that directly influence cultural variables, type three in figure 2.2.4, are *cultural instruments for environmental regulation*. Main examples are information and education. They require more complicated models than economic instruments, even in their simplest forms. Independent reasoning, research traditions and public spiritedness, to name a few diverse factors, influence the effects of, for example, public information on the environmental performance of a product in its life cycle. Their effects on the environment are always caused through changes in markets and technologies. These relate to processes that may be described in their material aspects. The latter only are at the level on which interference with the environment may be expressed. The model for predicting their effects will often take the form of rules of thumb, at least for the time being. "Environmental information will change consumer behaviour in those circumstances where the costs of a behavioral change are very limited" is one example. Cultural instruments, in turn, may apply more broadly than financial instruments and thus may be more macro in character. An example is environmental education.

4. Structural instruments
Instruments that directly influence structural variables, type four in figure 2.2.4, are *structural instruments for environmental regulation*. Main examples are changes in the rules governing property rights, such as liability rules and rules for acquiring land, and the building of institutions for environmental research. They may apply to nearly all processes in society. A single change in proof requirements for environmental liability directly effects decisions on any process that may cause environmental damages. Thus, structural instruments may be of a more macro nature than the other instrument types. Their relations with specific behaviour are often difficult to specify quantitatively. There will be an interrelation with cultural changes, which also affects financial economic variables, and through these, the material economy and the environment[1]. Their broad

[1] It should be remembered that structural model mechanisms here do not imply constancy. Although of course relatively stable over time, the structure of society develops autonomously and is changed through government actions for environmental reasons, for example. In the latter case the change is a structural instrument of environmental policy.

functioning may lead to the most fundamental long-term effects. The structural change from liberal democracy to communism in certain Eastern European countries after World War II, for example, had profound negative effects on economy and environment, as will hardly be denied. No one, however, can specify the quantitative difference of that forced structural change regarding specific emissions, for example of CFCs. Structural instruments of a less macro character are possible as well. An example is the obligatory procedures on decision-making in the firm, such as those affecting the health and safety of personnel, but which are now being developed for environmental aspects as well[1]. Another example are the property rights to trees grown in certain desert-prone public regions of China, now granted to local families.

Type of instrument and type of model are not related in a one-to-one manner. Effects of prohibitions, for example, may be modelled in the simplest way, in a mainly material model, but also in the most complex way, incorporating causal chains in the economy, culture and structure of society. At the other end of the spectrum, the effects of structural policy instruments will always require the four types of sub-models indicated. See figure 2.2.5 below for a survey. One peculiarity should be noted. The choice of the level of model is not a fixed one, given the fact that more complex, higher level models are largely lacking. For a fair comparison of different types of instruments, however, the model should incorporate the same types of variables. The simplest model, as often used to assess prohibiting instruments, is thus not useful in making a comparison of these prohibiting instruments with financial or cultural ones. The material economic sub-model may thus only be used to make comparisons between prohibitions[2]. Cultural instruments may be compared to prohibitions only by using models that at least include similar cultural and financial economic mechanisms, etc.

FIGURE 2.2.5 MINIMUM TYPE OF SOCIETAL MODEL REQUIRED, IN GREY, FOR THE MAIN TYPES OF INSTRUMENTS

	material economic (sub)model	financial economic (sub)model	cultural (sub)model	structural (sub)model
prohibiting instruments				
financial instruments				
cultural instruments				
structural instruments				

2.2.6 Models of the environment

Finally, the third system in the analysis, after government and society, is the environment, as influenced by material economic processes. The environment itself is a very complex dynamic system, with only some parts modelled. The environment may be defined narrowly, as the biotic and a-biotic surroundings of all social activities. External to both economy and environment (s.s.) there is the substrate of both (or of the

[1] The obligatory environmental audit now being prepared by the European Community seems to be an example.

[2] Material submodels of the economy may be used in the analysis of other instrument types only after specifying their effects in terms of the relevant non-material mechanisms.

environment alone ?). The substrate may be looked upon as 'the environment' of society and environment. Alternatively, substrate and environment together may be seen as making up the environment of society together. In systems terms, the substrate and the environment together form 'the environmental system' of government and the social system together[1]. However, the common, more restricted use of the term environment will generally be used here, differentiating between the environment as biosphere and the substrate as lithospere or geological system. Thus, ores untouched for geological ages belong to the substrate. Mining brings them into the economy where they are transformed. From there, eventually, they may go to the environment s.s. as emissions or back to the substrate, in some form of waste handling. In more formal terms, mining is an inflow from the geological system and emissions and waste, an outflow to the environment system.

The model of the environment should reflect what concerns there are about the environment. These revolve around three main value areas:
◇ human health
◇ functions of the environment in economic processes
◇ the value of nature per se.

See in this sense Udo de Haes (1992) and a more elaborate treatment of aims and principles in Part Three. As reflected in the now dominant concern for sustainability, the time horizon of the environmental model should be very distant, perhaps extending to hundreds or thousands of years. It seems an impossible task to model the effects of society on the environment in these terms, even if restricted to main lines only. Two examples may bring home the complexities involved.

Acid sulphur dioxide emissions are a good, suitably complicated example. An effect chain may be constructed on the basis of its properties as an acid, its substantial global cooling effects, its effects on long distance visibility, its health effects through the irritation of human mucous membranes, its corroding effects on most metals and limestone building materials, its structural contribution to CO_2 emissions through the increased weathering of limestone, its deposition in different types of area, its adverse effects on different agricultural crops, its adverse effects on forests, especially in non-buffered soils and its disastrous effects on non-buffered lakes where it kills all higher plants and animals. Directly, or through mechanisms further along the effect chain, sulphur emissions are thus related to all three major environmental value areas. Some items are stated in terms of the main value areas already, e.g. the irritation of human mucous membranes. For some of these effects the chain involving valuable elements is still extremely long. It is not possible now, nor will it be for a long time to come, to model all these effects together. The whole of ecology and the earth sciences are involved, as are much of physics and chemistry, and systems analysis. The results of these models, ideally, would be stated in terms directly related to the main values indicated. As part of the model, these normatively-based goal variables would still have to be specified, for the meaning of the detailed empirical analysis to be clear. Furthermore, the global cooling effect of SO_2 emissions can be assessed only in relation to the expected global warming emissions. As

[1] Difficulties will remain as long as there is an "outside". Behind the substrate is the inner part of the earth. Above the atmosphere extends space, with energy flowing in from the sun and other particles reaching us from still farther away and energy and hydrogen flowing out.

long as the latter remain dominant the cooling effect offsets global warming partially and thus is beneficial.

A second example is that of CO_2 emissions from the burning of fossil fuels. These may lead to global warming. That, in turn may lead to different climate changes at different places and to a rise in sea level. These relations are already highly disputed, for some scientists even as to the direction of the effects. Climate change is not a problem in itself. In the temperate and polar areas most people would see a modest increase in temperature even as an advantage. It is the system disruption caused that is reason for concern. Flooding and drought to a degree not experienced before would make obsolete investments and human which together with the effects of nature, have hitherto been productive. Sea level rise will require huge expenditures for fighting the seas in low-lying areas and many of these usually highly productive areas would be lost to the sea forever. These effects should in turn somehow be connected to health, to functions of the environment and to the value of nature. Some notions as to such relations of course do exist, but not much more than that. So how do we know what is relevant for environmental policy and how can this relevance be related to this interference?

The best strategy would appear to be to divide the model of environmental effects into two parts, a broadly acceptable quantitative one and a highly speculative, quite informal qualitative one. Usually, quantitative relations can only be substantiated for causal effects between a limited number of variables for a limited amount of time. When a point is reached in the causal chains where effects diverge to many other variables, the quantitative analysis stops short. From there on the mainly qualitative analysis takes over, linking the still quantifiable effects to the three main value areas. The effects of interferences that still can be quantified are *environmental problems*, with global warming, ozone layer depletion, acidification and the spreading of toxic substances as prime examples. With global warming, the analysis currently stops where the relative contributions of different substances to global warming are assessed. The amount of global warming expected, and all consequent effects, now belong to the realm of qualitative assessment, until a broader consensus arrives on further steps in the causal chains. If consensus forms - as seems to be happening with the quantification of the rise in temperature expected in the course of time - instruments and policies may then be focused more precisely on that then on a quantifiable problem further along the effect chain. Focusing on a limited number of problems instead of the high number of emissions and extractions, may reduce the complexity of policy development and analysis crucially. It also allows the development of more aggregate macro instruments, see below.

At present, the definition and modelling of some global environmental problems seem to lead to an astonishing convergence on main lines, at least for the global problems of ozone layer depletion and climate change. This convergence may easily break down as the results from Rio de Janeiro suggest. The climate problem, for example, may be defined as a CO_2 problem from fossil fuel resources. In that case the rich industrial countries are the greatest contributors, with the United States far in the lead, both per capita and in absolute terms. It may also be defined in relation to the global warming potential[1] of all substances contributing to global warming. Methane is then the main cause of global warming, contributing well over fifty percent to total global warming[2]. Main causes are paddy rice growing, animal husbandry with ruminants, coal mining and distribution losses of methane gas. Non-industrialized countries then are major sources of emissions, see the case on energy and global warming in Part Five. It is assumed here that a set of problems may be defined that broadly cover all purely non-local, non-site-specific effects on the three main value areas. Predominantly site-specific policies such as zoning and the forming and protection of nature reserves are not treated here. Effective macro policies may reduce the importance of these site-specific policies. Even then, site-specific environmental policy (not treated here) will remain a major area of political concern.

Cutting environmental reality into separate independent pieces associated with different problems is *not* necessarily based on the Descartian conviction that the total is the sum of its parts. It is the practical limits of intellectual complexity allowable in policy oriented analysis that puts a restraint on the broadness of environmental models. The limits imposed are always unsatisfactory in the sense that the richness and complexity of reality is reduced to a poor model. Extending the scope of environmental models, however, is unsatisfactory as well. Rational reasoning directed at specific policy instruments and policies then becomes impossible and broad acceptance of these complex models becomes nearly impossible.

Each aim, or problem, can be treated independently in the quantified part of the overall model. In the qualitative part of the model the problem may be connected to the three main environmental value areas. Global warming is connected to all three value areas. It will cost human lives, it will disrupt many production processes, and it will lead to the disappearance of many species. Thus problem variables are not problems in themselves. They can only be defined as such on the basis of the qualitative model; they are problems only because of their negative effects on the three value areas. This situation leads to severe problems in policy design as the relative importance of different problems can only be assessed qualitatively. Their quantitative trade-off could not be a matter of consensus, even if consensus on measures for and priorities of 'final' values existed.

Before the relative importance of problems can be assessed, eventually resulting in the assignment of priorities ("weights"), these problems first have to be defined carefully. At a general or generic level, chemical threats to health, global warming, ozone layer depletion, desertification, acidification, eutrophication, and species loss are main examples of problems that to some extent may be analyzed and solved independently. Problems may also be analyzed more specifically, however, in relation to the qualities of the site or sites affected. For purely local problems, such as the destruction of a given nature area or habitat, site-specific models will usually be required. But even global

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[1] The Intergovernmental Panel on Climate Change quite authoritatively quantifies GWPs for the main contributing substances. The model period used is usually one hundred years, but sometimes other periods are used as well, e.g. twenty years. The GWP_{100} of methane is 21, the GWP_{20} is 63, due to the shorter life span but stronger infrared-capturing qualities as compared to carbon dioxide.

[2] CFCs and similar compounds, now being phased out for ozone-depletion reasons, have been left out of the percentage computation. Their contribution still is subtantial but is declining rapidly.

problems, such as that of global warming, may be worked out in terms of a specific site as well. The models may then show what changes in temperature, rain fall, sea level, etc. may be expected for specific areas, and what the further effects will be. On the other hand, purely local effects, such as the destruction of one specific habitat, may be analyzed disregarding site-specific elements. Habitat destruction could be treated at a generic global level, in terms of the average species loss to be expected for example due to one degree Celsius of global warming.

Site-specific analysis, considered in its most extreme sense with all factors present in one location, precludes any movement towards macro instruments. Problems where aspects of location are decisive can therefore scarcely be treated with macro instruments. Abstracting from aspect of location "as they really are" does not mean, however, that only a global analysis is possible. The more generic analysis may relate to types of sites. Acid precipitation may be differentiated in a few classes as to the sensitivity of the soil to acids. But the analysis may still require that the location of interference be known. Macro-instruments directed at reduction of that interference cannot then be developed. The emphasis here will be on problems for which a generic treatment is possible, at the global level or differentiated as to a few types of sites. The analysis is thus restricted to problem types which may abstract from the specific location may be abstracted. This does not mean that a site specific analysis of such problems may not be worthwhile, or that policies directed at specific sites should be abandoned. That remains a matter of assessment, once the effectiveness of macro-instruments is established.

2.2.7 The society-environment interface: policy instruments 2

Society is connected to the environment at the material level only, the connection being the environmental interferences. It is the material aspects of society that may exert an influence. Hence the material economy is to be related to the quantifiable problems defined for the environment. The basic unit in the economy is the individual process, in its material aspects. The basic unit in the environment is not so clear. In the long run it is only problems that any interference can be linked to quantitatively. The smallest unit for the interface is one type of environmental interference caused by one economic process. Emissions of potentially harmful substances, the extraction of depletable resources, and encroachment of nature are the main categories into which this interference can be classified, see Udo de Haes (1992). As indicated above, this interference may be described by abstracting from the specific locations where it occurs and hence the modelling of its further effects cannot be site-specific. The question in this section is how a more aggregate interface may be built around this smallest interference, by aggregating more types of interferences than just one and more processes than just one.

Without such a systematic interface society - environment, policies would still have environmental effects, since any activity has smaller or larger environmental effects. These effects could be considered when evaluating policies. The instruments of such policies, however, would at best constitute instruments for environmental policy[1] more loosely defined (instruments sensu lato) than that developed here (instruments sensu

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[1] In the Netherlands, such a combined assessment of policies has been named integrated policy development, with the prime motive for policies with non-environmental areas of governmental concern. The importance of integrated policy analysis decreases if specific instruments for environmental policy analysis are available that apply to such situations.

stricto), if being instruments for environmental policy at all. Current discussions on instruments for energy policy, such as energy taxes, may be seen in these two different ways. Either they are part of broader policies, e.g. to acquire the *prime-mover* advantages of now introducing the energy price rise expected later. In that case there is no specific connection to any one environmental problem; all environmental effects resulting may be taken into account. The choice of what to tax may then be guided by expediency, e.g. transport fuels and electricity use. If seen primarily as an instrument for environmental policy, in the broad sense, the tax might still be the same, taking into account all effects on the environemt resulting. In both these instances there is not spcified interface society - environment in the instrument. Alternatively, the energy tax may be an indirect tax on CO_2 emissions, as an instrument for environmental policy sensu stricto[1]. Then the choice of interface is not so free. Then the emission concerned would be specified. Measured indirectly, it could be specifed in terms of the carbon content of fossil fuels only, with a refund for any carbon not emitted as in the case of permanent storage.

Here the analysis focuses on instruments for environmental policy sensu stricto, through specifying the material society-environment interface *in the instrument*. As stated before, the most basic interface for such an instrument is one process causing one relevant form of environmental interference. There would then be three different approaches to follow in turning the interface more macro. The first obvious option is to take more then one environmental interference, e.g. all heavy metals emitted instead of one, as the environmental object of the instrument.

The second option is to enlarge the interface at the side of the environment and to direct the instrument at some partial mechanism in the environment. There are a limited number of problem areas that can be specified now in environmental sub-models. Incorporating these problem sub-models into the instrument extends the interface to its maximum, at the envrionmental side. The appeal of this option is that a limited set of problem oriented instruments might cover all (supra-local) environmental problems. The number of different types of interferences (i.e. resources and emissions) is much higher. The problem oriented approach thus could be a step towards more macro instruments as compared to instruments for each interference, other elements of the instrument being equal. These two options will be treated together as one of the dimensions for aggregation.

The third option, not at the environment end but at the society end, is to aggregate individual processes into larger units, e.g. as all the processes constituting a firm or all the processes required for the functioning of a product.

These options for aggregation will be worked out in more detail in the next chapter. The discussion so far is depicted in figures 2.2.6 and 2.2.7 below.

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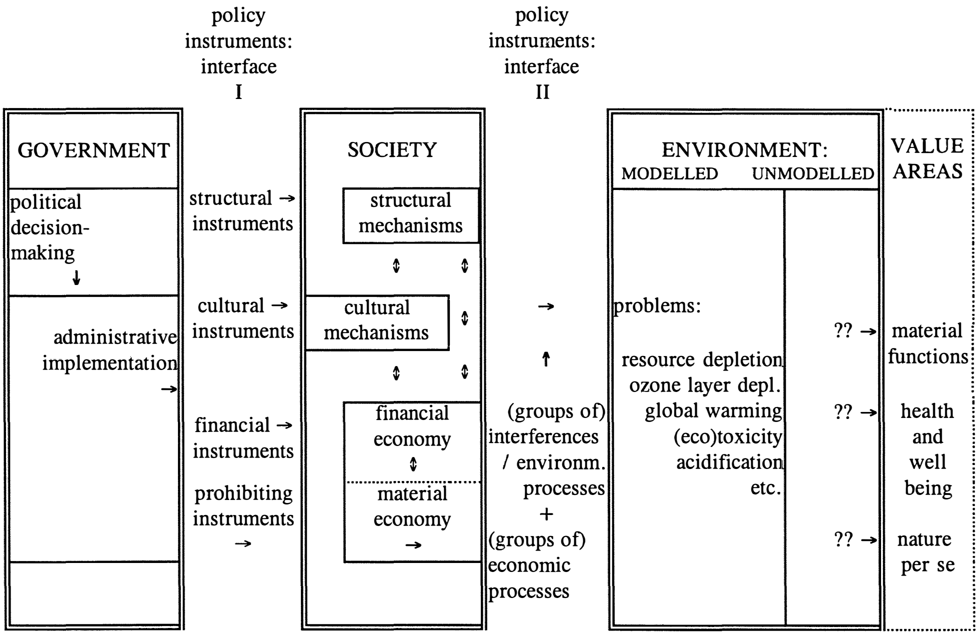
1 In a study for the Commission on Long Term Environmental policy (van Manen et al 1991) we distinguished four scenarios. One is the flexible- response scenario, worked out in this study as the most ideal but perhaps unfeasible strategy, using instruemnts for environmental policy sensu stricto. A second, is the spearhead scenario, concentrating on a limited number of environmentally relevant activities, using instruments for environmental policy sensu lato. Energy savings policy may be such a spearhead.

FIGURE 2.2.6 THE EFFECT CHAIN OF INSTRUMENTS FOR ENVIRONMENTAL POLICY AS DEVELOPED IN THIS CHAPTER.



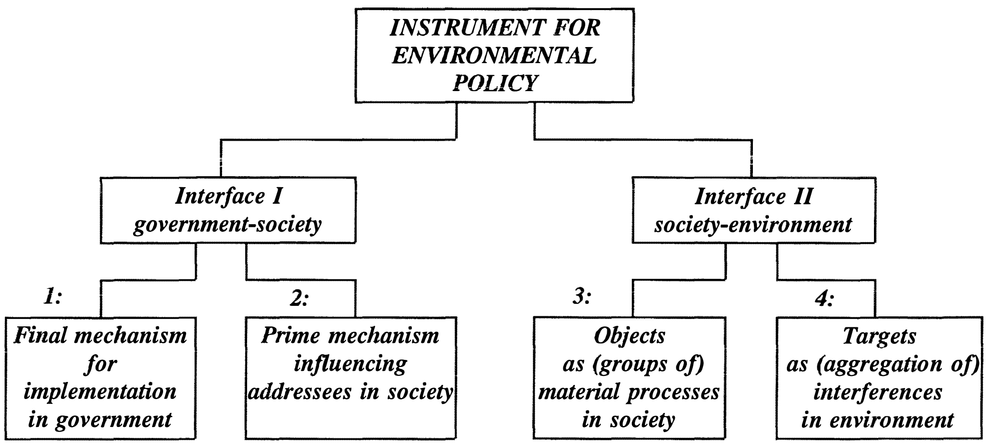
2.2.8 Conclusions

The function of models discussed here is to help define, assess and select instruments and policies. Modelling how decisions on environmental policy are made, as a political process, is therefore not relevant. This restricts the scope of the models substantially. Feedback loops from society and environment to government will thus be avoided in the analysis. The results of the analysis, if practically applied, will help shape the feedback loop. For more practical reasons, the feedback loop from environment to society can also be left out of models. Hence all feedback loops between the main system elements may be excluded, for both theoretical and practical reasons. The general model structure will thus be a simple, linear one. There are three sub-systems only, government, society and the environment. These are connected by two interfaces, stating causal relations in one direction only. The structure of the analysis is thus a very simple one. In each sub-system, complexity is unavoidable, especially in society and the environment. This model for instrument and policy analysis has a restricted domain. The following instruments are excluded:

◇ those developed *and* implemented in a continuous interplay between government and society, as in some neo-corporatist approaches to politics
◇ cases of direct intervention in the environment by government

The model of government may remain very limited, specifying only the implementation steps required for some instrument. The model of society should incorporate four different types of relations. They are material relations in the economy, financial relations in the economy, and, more general, cultural and structural relations. The model of the

FIGURE 2.2.7 FOUR ELEMENTS DEFINING AN INSTRUMENT FOR ENVIRONMENTAL POLICY (SENSU STRICTO)



qualitative part stipulates effects on three main value areas, human health; functions of the environment in production and consumption; and the independent value of nature. The quantitative part stipulates the effects of societal interference on intermediary variables in the environment. A number of such intermediary variables, covering the most important effects on the main value areas, are the *problems* environmental policy is meant to solve. The models assumed are not local and/or site-specific. Models of all three sub-systems are widely disputed and will remain so. An astonishing consensus exists on some main problems, especially the global concerns of ozone layer depletion and climate change.

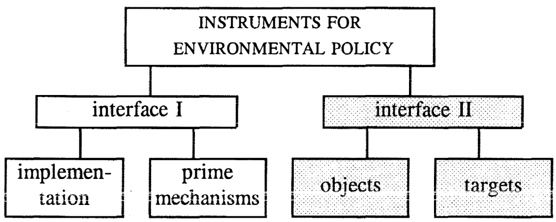
Instruments for environmental policy can be defined in terms of the two interfaces or, putting it the other way around, the instruments *are* the interfaces.
The interface between government and society is to specify the mechanism that sets in motion the social processes for environmental improvement. The four main types of relations in society define four main types of instruments: prohibiting, financial, cultural and structural instruments. Going from prohibiting to structural instruments opens up possibilities for more encompassing macro type instruments.
The interface between society and the environment consists of at least one economic process, in its material aspects, and one form of environmental interference. Possibilities for more macro type models of the interface may be designed by taking more then one individual example of interference. Two main possibilities exist: aggregating interferences into groups, ultimately in terms of problems, and grouping individual economic processes into larger units of related processes.

An instrument for environmental policy thus may be described in terms of at least four elements: at the government-society interface there are final *governmental implementation mechanisms* and *prime societal mechanisms* changing the behaviour of addressees as the direct subjects the policies relate to At the society-environment interface there are *groups* *of material processes, as objects*, with more or less aggregated *interferences in the environment as the targets* of the instrument. See figure 2.7.8.

There always is a relation between addressees and the object of the instrument. The relation may be a direct one. With individual permits, for example, the addressees are the owners of the process or process concerned (or their representatives). In that case the addressees are those responsible for process or groups of processes specified in the object of the instrument. The relation also may be more complicated, for example as with the ecolabel on a product based on problem oriented life cycle analysis. There, the addressee is a consumer making a choice between products, while most of the processes concerned having environmental effects are the responsibility of several producers and waste handlers.

These are the main outlines of the framework used for the analysis. The remaining chapters of Part Two and a fair portion of Part Four are devoted to working out the more macro type interfaces, i.e., the design of macro-instruments for environmental policy.

2.3 THE MATERIAL INTERFACE BETWEEN SOCIETY AND ENVIRONMENT: AGGREGATING INDIVIDUAL INTERFERENCES



2.3.1 Introduction

Instruments for environmental policy have been defined as the combination of the two interfaces in the model, one specifying the main social mechanism in implementation, the other somehow containing environmental interferences. It is assumed for the moment that these two elements, which form an instrument for environmental policy, may be varied independently. In this chapter several interfaces are specified between the material economy, as the material part of society, and the environment. The combination of an option at this interface with one option for policy implementation and its direct working mechanism in society, at the other interface, constitutes a full instrument. That latter step will be taken in the next chapter. Current policy instruments, especially installation permits and product standards, take one installation or one product as an object, and one technology or a number of substances as a target. The aim here is to define higher levels of aggregation that may be part of effective instruments. As a first step, the analysis aims at specifying the interface in a double way, as material economic processes related to environmental interferences. Stating the target of an instrument for environmental policy in terms of its environmental interferences is not a common practice currently. These instruments usually specify only technologies as non-environmental characteristics of products and installations. These characteristics are expected to induce environmental improvements in comparison to products and installations without them. Examples are the type of water purification to install in paper mills and the catalytic converter to install in cars. In a very strict sense these instruments would thus not be instruments for environmental policy. The specification of the interface will be directed only at instrument parts that centre around environmental interferences. Thus, the current policies on installations and products fit into the scheme only if described in terms of allowable emission levels, e.g. the amount of sulfur dioxide emitted in electricity production or the amount of NO_x emitted by cars (see type 1* in table 2.3.1 on p.51).

Any process in the material economy may influence the environment in a number of ways, through material[1] inputs extracted from the environment, both renewable and non-renewable; through material outputs emitted to the environment, i.e. polluting emissions of substances, energy and radiation; and through activities disturbing ecosystems and

[1] 'Material' as an adjective includes substances and energy. Material as a substantive is usually used in relation to some economic process. Milk is what a cow produces, gasoline is what a car consumes, paper is what you can write on. Each consists of substances and the potential energy contained in these.

landscapes. Each of these three types of interference may be related to all the three main value areas[1]. Extraction from the environment of the renewable resource of tropical wood, for example, is not only related to the depletion of that resource as a loss of environmental function; it also leads to a fall in the quality of the ecosystems involved. The climate change that may result endangers the health and life of millions of people.

There are two restrictions in the analysis here to reduce its complexity. The first is to disregard the disturbing type of interference, as matter of setting priorities in this study. Moreover, the effects of disturbances are primarily local, e.g. building a road either through a desert or a nature reserve[2]. Attention will be restricted here to extractions and emissions. How relevant the results will be for the disturbing interference is a question for later concern. The analysis now focuses on material flows of substances and the flows of different forms of energy.

The related second restriction is to the spatial level of the effects considered. The effects of emissions may be differentiated according to the location of their source, because transport mechanisms in the environment connect the source to specific targets at the boundary with the environment, and from there to further mechanisms in the environment, e.g. eutrophicating ammonia emissions that precipitate into an oligotrophic nature reserve. If transport in the environment is over larger distances, the location of sources becomes less relevant. This means that, depending on the type of problem, the purely situational aspects of emissions often need not be considered. With climate change for example, the location of a CO_2 source is not relevant to any practical measure[3]. However, in the defence of the last habitat of some oligotrophic plant species against only shortly airborne ammonia, the location of ammonia emitting animal husbandry is decisive[4]. In specific instances this local level may remain relevant. A more aggregated level of instrument, however, that abstracts from specific situational aspects, may still achieve a reasonable problem-solving level at any specific site. This would imply that at most locations the problem no longer exists. For the remaining locations still excessively adversely affected, there are two potential solutions. The general level of protection can be increased further, again through instruments with an aggregated society-environment interface. Or additional specific measures might be taken for the locations deserving extra

[1] See Udo de Haes 1992 for a treatment of the environment model in a similar sense. Opschoor and Reijnders 1991, p.16, similarly state three main types of environmental indicator: pollution, renewable, non-renewable and semi-renewable resources, and biological diversity. These three seem inhomogeneous as to the place in the effect chain. Pollution is at the level of interference, right at the start of the environmental effect chain, while biological diversity is quite at the end of the environmental effect chain.

[2] It is possible of course to abstract from locally differentiating characteristics and to assess disturbances as to some main types of situations. Adding a road in the countryside diminishes the average size of ecosystems. Effects of this kind might be modelled for one or more average situations, e.g. by MacArthur and Wilson's equilibrium theory of species diversity, the 'island theory'.

[3] It is possible to specify all effects further in the causal chain as to the specific locations, that is all locations, involved. In the global warming problem, for example, American models predict a very limited net effect for the United States, with advantages in some areas and disadvantages in others.

[4] This is the case only if overall pollution with NO_x is low enough to allow the further existence of oligotrophic plants. The incidence of ammonia emissions has been modelled as a support tool for situational decisions, see v.d. Voet and Udo de Haes 1989.

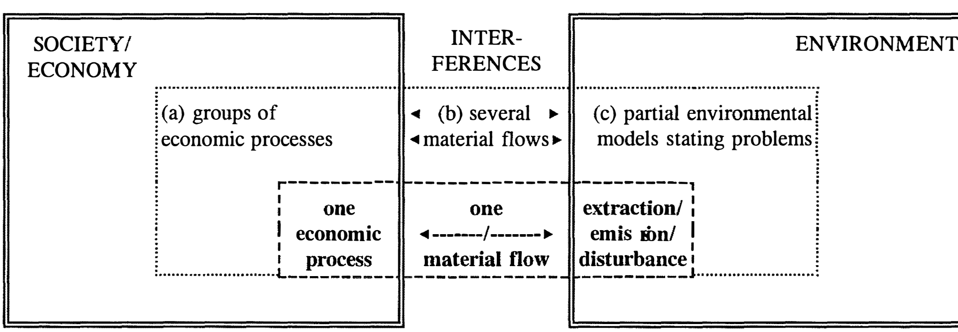
policy attention, e.g. zoning laws, individual permits and the like. Here, this study concerns the non-site-specific analysis of interferences[1] related to non-local problems.

The simplest interface connecting economic processes to environmental problems is one material economic process with one interference, e.g. one installation's emission of one substance during a certain period of time. See the dotted rectangle in figure 2.3.1 below. All further relations shaping that process and reacting to its change due to some environmental policy should be modelled in the societal model. All effects in the environment which finally affect some valued variables are covered in what is as yet an imperfect and primarily qualitative environmental model.

Elements for more aggregate interfaces may be generated in three different ways
(a) by taking together coherent groups of economic processes
(b) by adding several interferences
(c) by transforming interferences into intermediary environmental variables, e.g. problems, and thus adding them together.

See figure 2.3.1 for the place of these three elements in the causal chain. Each of the three elements may form a dimension for aggregation. Aggregating the interface, by grouping processes and by adding and transforming interferences, incorporates smaller or larger parts of the model of society and that of the environment respectively into the interface, as shown by the dotted rectangle in the figure.

FIGURE 2.3.1 ECONOMY-ENVIRONMENT INTERFACE: POSSIBILITIES FOR AGGREGATING THE ONE PROCESS - ONE SUBSTANCE INTERFACE

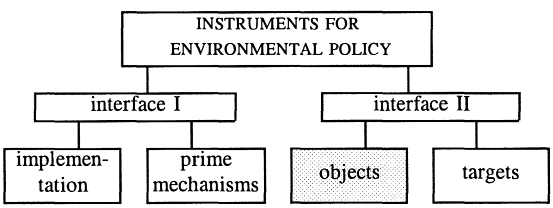


The main aim of the exercise in this chapter is to survey effective options for macro interfaces of instruments. Such options result from combining dimension a, groups of processes, with dimensions b and c, indicators of the environmental impacts of these processes. What is effective ultimately depends on the practical applicability of the resulting instruments. Of course, these instruments should have interfaces that are more macro than the current instruments for environmental policy.

[1] The Environmental Impact Statement is a main site specific type of environmental analysis. It is mainly used to support decisions on projects at a specific location.

The three dimensions related to the aggregation of this interface will be examined in turn. section 2.3.2 examines possibilities for grouping processes as the objects of policy instruments. The aggregation of interferences and their transformation into environmental effects, as the targets of policy instruments, will be then treated twice. First, section 2.3.3 will survey proposals from the literature, mainly commenting on the economic evaluation of environmental effects as one approach to aggregation. section 2.3.4 specifies the scale that will be used for ordering targets. Adding interferences and transforming them, though quite different operations by nature, will be treated there as a continuum, effectively merging the two scales b and c. The two scales remaining, "groups of processes", the objects of instruments, with "aggregated environmental effects", their targets[1], together form the society-environment interface created by instruments for environmental policy. Next, some promising combinations of these objects and targets will be selected in section 2.3.5. Certain targets of other instruments are then specified, both further and less far along the effect chain, and these are related to the potential efficiency of instruments. The chapter ends with conclusions, in 2.3.6.

2.3.2 Aggregating economic processes: a basic scale



One process, related to at least one interference, is the most basic unit for the economic object at the society-environment interface. The process may be described in its input-output aspects, as a system, or as a physical entity, as a product or installation. If several of these basic processes have been defined, these may be grouped into meaningful aggregates, as the objects of the society-environment interface. Such aggregates may have

- ◇ a material or technical economic meaning, related to what processes may be treated together for purely technical reasons
- ◇ a symbolic economic meaning, that encompasses processes on the basis of the function or value they have in society
- ◇ an environmental meaning, related to the type of interference.

An economic process may be described in several different ways. It may first be defined physically, as a material installation or product; it may be defined as a system that processes input types and volumes into output types and volumes; and it may be described in terms of its functioning for a certain period of time, as a producer of services; and finally the process may be characterized as a producer of values, ultimately described in monetary terms. Aggregates of these economic processes may be formed in the three directions indicated, starting at basic installations or products, see number 1* in the list of table 2.3.1.

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[1] The term target in no way implies the setting of quantitative norms, e.g. for emissions, concentrations in the environment, or problem contributions. In some instruments such quantitative norms may be used, as in emission permits.

Which basic types of aggregation will be taken into account in defining instruments? With the technical economic meaning as a basis for aggregation, there seems to be one option only. That is to take together all processes that together form a *technical unity*, see option 2* in the table. This option for aggregation assumes a definition of the smallest installation. Any installation, including products, can be considered as disconnected into parts. A car may be considered dismantled into an exhaust, motor, wheels, bearings, and several other parts. In that case it is the exhaust that produces the emissions. There is a limit in the further dismantling of parts where the analysis ceases to make sense. At that point the element can no longer be termed an installation. However, it does not matter much how the "smallest process" is defined, since aggregation to some meaningful level will always take place, at least in the technically connected installations of level 2*. At the other end, the technical unity might even include installations at different locations. Although exact boundaries are difficult to define it is clear that a paint spray installation with a compressor constitutes a technically more cohesive unit than the combination of an oil refinery and oil-based electricity production. It makes the most sense to define technical unity to imply installations at the same location. The main instrument example using this first step in aggregation is the installation permit common in all industrialized countries.

Next, a number of miscellaneous options for the aggregation of objects are used. There are two options are related to *geographical units*. First, all processes at one *location* may be treated as one unit[1], see option 3* in the table. One location is defined, according to common usage, as one area, used by one firm, for one or for more processes. Only fixed, as opposed to mobile installations and products, may be aggregated this way. Secondly, all processes in some larger *region* may be treated as one, including those of different owners, see option 6* in the table. Tradable emission permits, with trade restricted to a region, are an example[2]. Zoning laws would also apply at this level.

Another option, number 4*, is related to *ownership*. All installations owned by one owner may be treated together as well as those at different locations and areas. See option four in the table below. This is quite usual in taxation and in the financial analyses carried out for the many owners of corporate bodies. In environmental policy the environmental audit of the firm might be at this level. Industrial firms such as General Electric Plastics, DOW Chemicals and Monsanto, and even a software house such as BSO have surveyed all environmental effects caused by their operations and have published the results in the form of additional information in their annual financial accounts. Another example is Maximum Average Fuel Economy of all cars sold by one manufacturer or importer, now obligatory in the US.

The next option is to take together all processes of a *similar type*, i.e. belonging to one *sector*, option 5* in the table. Such groupings seem highly arbitrary since specific definitions always create overlapping categories. Steel production co-produces cement.

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[1] Of course, implementation is always organized on a geographical basis in terms of the administrative body concerned. That is not the subject here.

[2] In the example of tradable emission permits in the US, it should be noted that the amount of emissions allowable are not measured, usually, but are translated back into technical standards in permits for technically connected installations of type 1* or 2*.

The bulk output of certain medicine production is fodder. Furthermore, no one is specifically responsible for all processes in a sector. As in the regional approach, a responsible body could be formed, e.g. a representative body with certain power over its members. In the Netherlands, the "Foundation for Packaging and the Environment" has acquired that position. It is the partner in a covenant with the Dutch Ministry of the Environment (Convenant Verpakkingen 1991) aimed at curbing the amount of packaging waste. This corporatist type of policy development and implementation is now being tried out in the European Community, see EC Manual (1990) on Priority Waste Streams. Corporate taxes were set up according to this model in the Peoples Republic of China when it came to power. The government defined sectors and fixed an amount of tax to be paid by the sector body. The bodies being formed were representatives of the then still private firms. They were free in deciding how to distribute the taxing burden amongst the firms concerned.

Several options, 7* to 12*, are based on the *functions* that processes may have. A first option is related to the *prime function* of a process, i.e. its main economic output. Emissions could thus be expressed per *unit of output of one process*, such as a product or service, see option 7* in the table. A general rule could stipulate the level of an emission, not specifically for one installation but for all installations producing such an output. Take for example electricity as the prime product of an electricity generating system. In Germany, the *TA-Luft* specifies the maximum amount of sulfur dioxide per kWh of electricity produced. Another example is the minimum level of energetic efficiency for a number of household appliances now considered by the European Commission as a means for reducing CO_2 emissions. Only one process is considered here.
Secondly, all processes in the *production column* of a product might together be seen as one unit, see option 8* in the table. No operational examples exist. Several studies link materials production to environmental problems. The use of aluminum in drinking cans, for example, has been the subject of environmental policy because of its production energy requirements. A more extensive Dutch study on materials goes one step further and includes in the analysis the waste treatment of the materials, after their functional use. This leads to the problem that all applications, with their specific methods of waste treatment, must be known. Extending the analysis to the functional uses of the material results in the life cycle analysis (LCA) of the products the material is used in, see the next option.
Thirdly, there is the *product*[1] *life cycle*, as the combination of all processes required for the functioning of a product, see option 9* in the table. These processes relate to resource extraction, all production processes, with the handling of their wastes, all processes related to the functional use of the product with their wastes, such as petrol production and the maintenance required for driving a car, and the handling of the discarded product, including all types of recycling and waste processing. The life cycle analysis is very crudely operational, as in the German ecolabeling system. Life cycle analysis is developing rapidly. One problem to be solved is that there may be several functions that go together, as with milk and wool from a sheep. The problems of defining one functional unit, and of attributing part of all related processes to that one functional unit, will be dealt with extensively in Part Four.

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[1] There seems to be no difference here between an intermediary product and a production installation.

There is one special element in the life cycle analysis of products. The function of a product as specified in this analysis is a prime indicator of its value. It would allow a first assessment of negative environmental effects against the positive value a product has, systematically within the instrument. Based on the life cycle analysis of products there is a whole range of further aggregations.
Fourthly, there is the *life cycle of groups of products*, option 10* in the table. These groupings may be based on a very broadly defined function, requiring the use of several products. Different ways to spend a fortnight's holiday in Greece might be an example. Comparisons may be made between even more abstract functions that together form some unit. The life cycle analysis of different ways to spend holidays in general, and of consuming biological dynamic foods instead of ordinary foods are examples. Thus, life styles can be analyzed in terms of their environmental targets. The basis for such an analysis is the life cycle of individual products. Since decisions on such groups of products in consumption, production and policy formation imply direct action for all products involved, this option for interface analysis may be seen as a tool for *strategy development*, influencing later choices on individual products.
Fifthly, there is the *life cycle analysis of one ECU's worth of consumption,* option 11* in the table. It abstracts from specific products and generalizes their functions into values, in monetary terms. The analysis may compare any two products, or a ranked average of several products, according to the amount of the interference caused by each ECU spent. Given the fact that people spend a certain amount of their income, and that their income is mainly independent from the purchases made by individuals, spending income on purchases having the lowest environmental impacts per ECU spent helps minimize all environmental problems related to production, consumption and waste handling. The basis for this analysis is the broad availability of life cycle studies on individual products.
Sixthly, there is the *life cycle of all final products together*, option 12* in the table. Final products are products that themselves are consumed, instead of being applied in another production or waste handling process. Consumption here includes public consumption, as for recreational facilities. The processes involved in all life cycles, by definition, together cause all environmental problems, including those related to depletion and ecosystem degradation. There are no actual instruments operational at this level. Curbing population growth and curbing consumption per head of the population, both proposed by environmentalists, are the demand-side mechanism here that could be relevant for environmental policy.

The other side of the same coin, at the global level, is the sum total of *all material processes* in society. Together these also cause all environmental problems. They are the same processes as in the life cycle of all final products, but seen from a different point of view. The difference is very similar to that between national income and national product. Restricting the proportion of the working population and restricting their working hours are the supply side mechanisms that might be relevant for environmental policy.

At a global level, all consumptive activities together require the functioning of all material economic processes. In that case the options 12* and 13* fully coincide[1]. However, at the

[1] One difference is that the consumption in one year requires processes spread out through many years, both in production and in waste handling.

TABLE 2.3.1 POSSIBILITIES FOR AGGREGATION OF OBJECTS AT THE INTERFACE SOCIETY-ENVIRONMENT[1]

Technologies

1* *Basic installation or product*

The functioning of one basic installation constitutes the basic element on which all practical policies are built. Most basic installations are composed of separately produced parts.

2* *Technically connected installations*

Several basic installations may together form the technical unit that is defined in some applied instrument.

Miscellaneous

3* *Location*

All processes functioning in one location may be analyzed together. Examples are the Dutch integral environmental permit and parts of the US "bubble" under the Clean Air Act.

4* *Ownership*

All activities of one firm may be analyzed together, as in environmental audits conducted at this level.

5* *Sector*

All processes taking place in one sector may be taken together as the unit for some instrument. No examples are available.

6* *Region*

All processes in one region may be brought under the same instrument. No examples exist. Tradable emission permits in the US (banking and netting together) go in that direction.

Functions

7* *One unit of output of a process*

Specific installations are abstracted from; only their main output is related to the environmental interference.

8* *Production column of product*

All processes in the production column required for the production of a certain intermediary or final product. May be seen as part of the life cycle analysis.

9* *Life cycle of one product*

The life cycle of a product consists of all processes that are required for the function of the product, in the amounts necessary.

10* *Life cycle of groups of products*

Some functions require several products. Going on holidays, for example, requires airplanes, hotels, food and drinks. These may be analyzed together.

11* *life cycle analysis of one ECU's worth of consumption*

Specific ways of spending an ECU could be compared in terms of the interference caused.

12* *Life cycle for all final products together*

This is the total of all interference associated with some level of national income.

All technologies

13* *All material processes together*

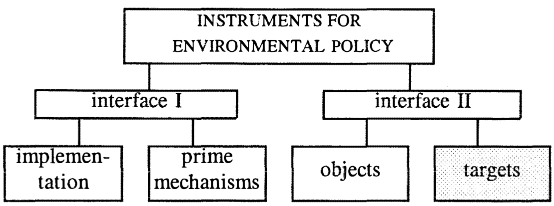
At a global level, this grouping is identical to number 12*, the difference being the route of aggregation. At a national level, there is a substantial difference to twelve, since the production in one country differs sharply from its consumption.

national level this is not true. National production is partly based on import and is partly exported, while a substantial part of all products consumed is produced abroad. Thus at the national level option 12*, the total of all life cycles, indicates the processes and their environmental effects caused globally, while the total of all material processes in the country indicates the processes for which a country may be held responsible directly.

[1] This classification builds on Huppes (1990b).

The order of the aggregates of processes, as in table 2.3.1, indicates an increasing level of aggregation, grosso modo. The underlying dimensions used are type of technology, spatial scale level, ownership and function. Together they form a typology of aggregates of economic processes. It constitutes the first basic ordinal scale for aggregating processes at the society-environment interface. Addition of interferences and their modelled aggregation in the environment together form the second basic scale with which to generate the full interface. That subject is treated in the next two sections.

2.3.3 Aggregating interferences: the economic approach



The current economic literature on instruments for environmental policy is dominated by the notion that financial instruments may reflect the negative value of the adverse effects caused, for example by an emission. A vast body of literature has accumulated based on the quantification of the marginal effects of policy instruments in terms of costs incurred and damages prevented. These marginal effects on costs, expressed as the excess of social costs over private costs, have the function to compare and further develop instruments, in principle leading to practical policies. These endeavours are in the main research programme formulated by Pigou in the Twenties and Thirties. Pigou proposed to internalize the external effects of consumption and production with a tax equal to the social (including environmental) damage done. For half a century a discussion ensued on the exact conditions under which advantages would or would not accrue. No practical proposals for such taxes had, or have been developed however, see the survey of Bohm and Russell (1985). The problem in developing the tax is not that of the instrument mechanism but of stating the value of the environmental effects of emissions. If that were possible, the Pigouvian taxes would become specifiable. If the quantification were available, each interference could be transformed into one general unit, giving the value of its environmental damage. Any unit of any type of interference could thus be transformed and added to any level of aggregation as desired. This attractive option has not yet materialized.

The position on both social and environmental modelling developed in chapter two does not allow the quantification implied. There are some models for quantifying effects of interferences in terms of problems such as global warming and acidification. For such important areas as potential health effects and resource depletion even these partial models are not available. But this is not all. There are no conceptual models available, and certainly no quantified empirical models, that describe valued effects in physical terms, such as the quality of nature, functions of the environment in production and consumption, and human health and physical well-being. A still further problem is that these valued effects cannot be known precisely enough to be valued in monetary terms. Since models for effects in terms of the main value areas are primarily lacking, and completely lacking for supra-local environmental problems, the dearth of practical proposals for the Pigou tax comes as no surprise[1].

The first recognition of this state of affairs is a famous article Baumol and Oates (1971), commenting on the proposals made by Pigou nearly half a century before[2]. Baumol proposed a simplification that could make the emission tax into a practical option. Instead of quantifying the environmental damage in monetary terms, a public decision would be made on the acceptable level of total emissions, as a collective standard[3]. Simple econometric analysis could then fix the level of the tax required for the complementary emission reduction[4]. Their proposals would reduce the environmental part of the interface again to simple interferences, such as the emission of a certain substance. The Baumol tax remains a tax on one process with one interference, with an equal level for all processes.

In an equally famous political decision a decennium after the proposals of Baumol and Oates, President Reagan's "Executive order 12291" (february 1981) stated that environmental regulations should only be enacted if it can be shown that the monetary proceeds will outweigh the costs of these measures. In line with Pigou, and now also tuned to the stated political demands, the discussion on ideal economic instruments went on. Methods for quantifying proceeds and costs were further developed. Nash and Bowers (1988) and Hoevenagels and Opschoor (1990) give a survey of both the literature and of the valuation techniques developed. It seems that most techniques (either based on WTP, willingness to pay, or CV, compensating variation) relate to local differences in amenities, perceived pollution and in other types of differences in current environmental quality. Hoevenagels and Opschoor (1990) state that monetary valuation is most sensibly applied where current states of the environment have to be compared with known, small and reversible differences (p.70). For comparing future states, and current large-scale, irreversible changes that are only partially known, the methods are not applicable.

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[1] The situation may be compared to the monetary valuation of a house, based on knowledge about the amount of some, but not all building materials, but without knowledge of architect, builder, location and size. The environmental valuation then also has the added complication that for the main effects, take species extinction, there are no markets possible to indicate their value. In a report of Resources for the Future, DeWitt et al. 1991, inadvertently, draw attention to some problems to this approach, which not only quantifies environmental effects but also economic ones. In their discussion on energy taxes, they propose a quantification of the economic effects of the tax. These include the consumptive value forgone because of higher prices, differentiating between consumptive applications of energy and types of energy. The results are the basis for a differentiation of taxes between energy sources, to minimize the consumptive value lost. The same reasoning makes them advocate research into a regional differentiation of energy taxes, as in different regions of the US different energy resources are used in different amounts in different activities, thus showing different levels of social costs per region. Such a differentiating approach would delay any new environmental instruments forever, taxing or otherwise.

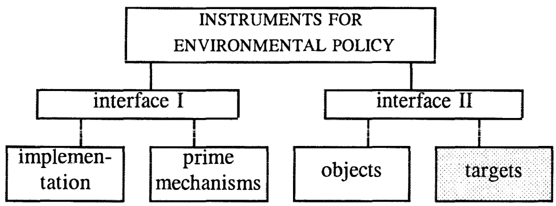
[2] A detailed study on the practical possibilities for monetary valuation in the neoclassical tradition, in Dutch is by Opschoor 1974, with a survey of results on pp.165 and following. Together with another author he surveyed the attempts at quantification more recently again, see Hoevenagels and Opschoor 1990. The conclusions remained the same, although much progress has been made. Quantification is possible to a very limited extent, and fraught with theoretical problems. Main elements cannot be quantified at all.

[3] The approach advocated by Baumol and Oates, setting prices to achieve a certain standard, still specifies more than is strictly required. Generally, indications such as "a substantial reduction in emissions" is the highest precision in environmental aim that can be argued. Usually, however, this vague description of the aim gives enough precision for reasonably argued quantification of policy instruments.

[4] Simple only if disregarding dynamic effects.

This study focuses on the design of macro-instruments for non-local problems. Here, monetary valuation cannot play a significant role now or in the foreseeable future, if ever.

2.3.4 Aggregating interferences: a combined environmental scale



One interference, related to at least one economic process, is the most basic environmental unit at the society-environment interface. For the aggregation of interferences there are two different approaches. One uses only the physical characteristics of the interferences, the other enlarges the interference through the modelling of processes in the environment set in motion by it. In both cases, the starting point is basic individual interferences. This "basic interference" is difficult to define, as was the case with "basic processes". For resource extraction, is it one unit of ore such as "iron ore"? Is it an ore within classes of concentrations of the main element? Should these be subdivided even further, e.g. according to other minerals present in the ore and to its exact location as an indicator of mining costs? With emissions, is it one substance emitted to water, air or soil, such as "oil"? Is it specific types of compounds? Or are isomers and isotopes to be differentiated? Interferences, both inputs from the environment and outputs to the environment, consist of flows of matter and energy in different forms. Given that these basic units are defined somehow, as materials, radiation, other types of waves, etc., there are several options for aggregation. I begin with those based on physical characteristics and then go on to those based on environmental modelling.

Physical aggregation

The first and simplest option in the physical approach to aggregation is incorporate several interferences of one process into the instrument. A permit, for example, may specify allowable emission levels independently for all substances relevant to the installation. *All substances* then are *treated parallel*, that is option b* in table 2.3.2. This, however, is fully equivalent to applying the same instrument to each interference consecutively. It does not appear to be an option that increases the macro character of the interface very much. It does, however, allow a simpler procedure of implementation. A cross-media approach, now the aim in issue of permits in most Western countries, may be implied by this aggregation in the sense that emissions to water, air, and soil are treated in the same procedure.

The second option for physical aggregation is to choose one *group of chemically similar substances* and treat these as a single entity in the operation of the instrument, that is option c*. An emission tax, for example, could apply to all volatile chlorinated compounds. These could be brought under the same heading based on some physical characteristic. That characteristic might be their combustion value, their molecular weight, the number of chorine atoms, etc. The difficulty here is that at the physical level there is no specific reason to choose between such alternative characteristics. An alternative may be defended only in relation to an environmental problem. The characteristic then is the central element in a problem module related to a group of substances, that is option f*. The combustion value is relevant for energy depletion, the number of chlorine atoms might be a very rough indicator of ozone depletion contribution.

The conclusion here is that in the choice of basic units some aggregation is always involved. Further physical aggregation leads to aggregates that are not related to the further effects in terms of problems they may have. Thus, such a purely physical aggregation, not related to problem contributions of the substance, cannot be an important option for aggregation.

Modelled aggregation: the problem approach

The second way of aggregating interferences is by transforming them with some quantified partial *problem module* from the larger primarily qualitative environmental model. This module, consisting of one or more environmental processes, then may be incorporated into the interface. These modules may be related to environmental values more easily than individual interferences. The problem modules specify the mechanisms influenced by environmental interferences to that point in the effect chain where effects are quite undisputed on the one hand, but are certainly relevant on the other hand, because of as yet non-quantified subsequent steps in the effect chain. These further steps may be quantified but then are widely disputed. Or they are only qualitative, or only consist of very general ideas. Global warming, for example, may be expressed in equivalent units of contribution to global warming. The life span of methane in the atmosphere is quite undisputed compared to that of CO_2. Moreover, the infrared retaining properties of both substances are undisputed. Fed into the "best" climate model, the best estimate of their relative global warming potential results. The best model for the prediction of these comparative effects is a dynamic item, scientifically, but at any moment there is a reasonable consensus as to what constitutes a 'good model'. Thus the relative global warming potential of these two substances may be undisputedly assessed. This can be done *irrespective* of the absolute amount of global warming caused, still a hotly disputed scientific subject. The aggregation is thus also independent of primarily unknown damage to human health, independent of virtually unpredictable long-term changes in proceeds from the environment, and independent of difficult-to-assess losses of quality of the environment itself. Together, these further effect chains form the main model of the environment. As with the model of society, this large and complex main model of the environment is virtually unspecified and where specified is widely disputed. At present further effects of problems can thus be assessed and evaluated only subjectively.

For filling in each module, the "equivalent unit" approach seems most apt. It will be worked out in some detail in Part Four. In that approach, an emission or extraction is translated into a contribution to some intermediary variable that, only in terms of the modules chosen, is the goal variable in terms of which environmental policies may be assessed quantitatively. The contribution to these *derived goal variables* is relative, as in global warming potential of some substance, e.g. methane, related to that of a reference substance, e.g. CO_2. Sometimes now the absolute contributions are given, as in the amount of acid emitted. however, 'moles H^+' could here be seen as the reference. The structure could be similar for each module. It would translate different emissions into one target variable, by first taking into account the decreased practical availability over the course of time through transport, binding and chemical decomposition, and secondly by its resulting effect on the derived goal variable. Indirect effects on the goal variable, as through metabolites formed and interactions with other substances, are preferably included as well. Generating the modules that relate interferences to problems has only begun in the last few years. The climate problem and the global warming problem are now well established. Other modules are being developed. See Guinée (1991) for a survey. Several options for aggregation (options d* to h* in table 2.3.2) are based on the assumption that the relevant problem modules will become available. The number of problem modules, for anything other than purely local problems, does not seem to be large. Not more than thirty problem modules would cover all main problems currently discussed, see Part Four.

The ultimate form of aggregation that now can be reached is into the contribution of interferences, through their contribution to problems, to a highly abstract unit called the "total environmental problem". The goal variables (equivalent to target variables or valued end-points) of each module have a derived normative meaning, because of their assumed contributions to the three main value areas. These contributions might be modelled quantitatively on the basis of models that, alas, are not yet available. Subjective assessments could fill the void between accepted but as yet "non-relevant" facts modelled in the module and the values it relates to. This is a procedure very similar to that using subjective probabilities instead of unknown risks. A difference with the latter procedure would be that it requires no evaluation. The subjective normative assessment may be individual or collective. A major difference with the economic assessment is that the latter suggests impartial results, while an explicitly ethical personal or collective political decision reflects the status of the problem as being of a quite different subjective nature. Furthermore, the scientific status remains very different since "problems" are assessed scientifically, while effects on the total environment are assessed ethically and politically. A final difference is that technically, the total environmental effect cannot be translated directly in terms of monetary proceeds and costs; that requires a separate step still to be taken.

In the context of democratically used policy instruments the subjective assessment could only be based on a collective decision. It translates individual problems into their contributions to a weighed total of all problems. The "scientific" alternative, the modelled generation of choices, would relate first to a subjective assessment of facts, then to their implications for different value areas, and finally to the relative importance of these different value areas. This now seems an insurmountable task[1]. Practical policy choices, however, demand the results of such a procedure if minimum requirements for rationality are to be fulfilled. The primary requirement is that the ranking of policy priorities is transitive[2]. Or, starting at the other end, if minimum requirements on rationality are

[1] It should be noted that those advocating an economic evaluation of environmental effects not only assume that such an aggregation is possible. They also assume that weighting is possible in monetary units that allow a further weighting against the preferences for private products. And they assume that these steps do not require subjective political decisions.

[2] See Sen (1969) for an authoritative full treatment of this subject.

fulfilled in practical choices made, there is at least one set of weights on the problems involved which can reproduces the choices made in a formalised manner[1]. These weights specify the contributions of each problem to the imaginary "total environmental problem". With it, consistent policy choices could be reconstructed. Whatever route is taken, the result would be a weighing of all relevant problems, as they are defined in the modules, into the total problem. The interferences of one group of processes could then be translated, in several steps, into their contribution to one total environmental problem. The two most aggregated options (options i and j in table 2.3.2) are based on this relative ranking of environmental problems. A summary of the options for the modelled aggregation distinguished now follow.

The minimal option in the problem oriented approach is where *one interference* is related to *only one environmental problem*, that is d*. There is not much practical difference then with no aggregation at all, as in a*. Here, the link to some environmental effect is specified quantitatively. Further aggregation is readily possible however, through the addition of problem scores of different interferences, e.g. substances.
The second option is to relate *one interference* to *several environmental problems*, that is option e*. The main difference from a* is that here too the relation to problems is quantified. There is no aggregation over substances, however; only one substance is regulated in the resulting instrument.
A third option is the aggregation of *a group of substances according to their contribution to one environmental problem*, that is option f*. The restriction is that only similar substances, in a material sense treated above under option c*, are translated into problem terms. Major existing examples are dioxins and PAHCs[2]. The dioxin group consists of several compounds and isomers of compounds. These are transformed customarily into a "standard dioxin", according to their quantitative contribution to a mechanism hazardous to health. Similarly, several PAHCs are translated into one standard PAHC based on their relative health hazards. Another example is the grouping of all cadmium compounds. These may differ strongly in their specific effects. With some exceptions, however, these compounds may quite easily transform into each other in the environment. Here all compounds add to a single "cadmium score" on a per cadmium atom basis. This group might be enlarged to "all heavy metals". Then a translation into some health threatening aspect is required, as now is done with dioxin and PAHCs. If some media-specific transport model is incorporated into an assessment of relative contribution, these groups may be aggregated over all the environmental media they are emitted to.
A fourth problem oriented option is to aggregate *all substances according to their contribution to one problem.* This forms the basis for purely one-problem oriented instruments, e.g. a global warming preventing instrument or an acidification preventing instrument. If all problems could be covered by one-problem oriented instruments, a very transparent instrument structure would result.
A fifth problem oriented option is to aggregate *all substances according to their contributions to every environmental problem.* This requires a "complete" list of problem modules. If such a list exists, and instruments formed are applicable, then this option is

[1] This approach is similar to that of von Neumann and Morgenstern 1943 concerning the utility of outcomes of choices. Utility there is not a "real" entity but a set of weights reconstructing a series of consistent decisions.
[2] Polycyclic Aromatic HydroCarbons.

TABLE 2.3.2 POSSIBILITIES FOR ENVIRONMENTAL AGGREGATION OF THE TARGET AT THE SOCIETY-ENVIRONMENT INTERFACE

No aggregation	
(α*	*Technologies expected to be relatively beneficial environmentally*) Encompasses the bulk of current instruments for environmental policies.
a*	*Basic substance per medium.* In environmental regulation several thousand individual substances may now be distinguished, subdivided according to the medium into which they are emitted. There are no standardized categories for resource extraction.
Physical aggregation	
b*	*All substances treated parallel* No aggregation, additive or otherwise, is involved. A much simpler administrative procedure is made possible, however, at the implementation interface of the instrument.
c*	*One group of physically similar substances.* It has proven difficult to single out one physical characteristic as the basis for aggregation.
Modelled aggregation	
d*	*One substance related to one environmental problem.* Very similar to a*.
e*	*One substance according to its contribution to every environmental problem.* Practically very similar to a*. Theoretically quite advanced.
f*	*One group of similar substances according to their contribution to one environmental problem.* major existing examples are dioxin and PAHCs, translated into a standard unit based on quantified health hazards.
g*	*All substances according to their contribution to one environmental problem.* This grouping abstracts from any specific chemical composition of interferences. "All global warming substances" is an existing example.
h*	*All substances according to their contributions to every environmental problem.* The analytical methods for such a type of aggregation are now being developed in life cycle analysis. There is no consensus yet on lists of environmental problems.
i*	*One substance according to its contribution to the "total environmental problem".* A political decision on the ranking of problems is required.
j*	*All substances according to their contribution the "total environmental problem".* Environmentally based choices between functionally equivalent product alternatives generally feasible. Required for assessing cost-effectiveness of environmental policies.

fully equivalent to the consecutive treatment of each problem with the one-problem oriented option g*.

The last two options differ from the preceding in that they require the relative weighing of problems. For a given list of problems this involves no technically complex procedures, a set of weights is enough. Experience in decision theory has shown that groups converge quite easily on normative weights attributed to problems. See Part Four for a further elaboration of the ranking of problems. The first option, i*, is directed at *one substance according to its contribution to the total environmental problem*, that is option j. No aggregation of substances takes place. If that option is possible, however, the step to the highest level of aggregation possible can easily be made.
The highest level of aggregation is where *all substances* can be incorporated into an interface *according to their contributions to the total environmental problem*. Based on this aggregation, any change in one process or in a group of processes could be

transformed into a general environmental evaluation score. With this option, technical measures for environmental improvement that score divergently on several problems may be compared. An example may show the practical importance of this seemingly abstract ranking procedure. One desulfurisation technique for sulfur-containing oils requires eight tonnes of carbon emissions, per tonne of sulfur dioxide emission to air prevented[1]. Is that technique worthwhile environmentally? That question may now paralyse policies since no authoritative answer is given. If policies proceed, an implicit answer is given, based on the subjective ranking by some environmental official. For preventive measures, as in design decisions in industry, the answer to this collective normative question is required. It should be available publicly and backed up by authority, in the form of a set of collectively decided normative weights.

2.3.5 Promising combinations for instrument interfaces

With the results of the preceding analysis, a large number of aggregated interfaces at the society-environment boundary could now be defined. Each possibility on the dimension of process groupings, *the object of the interface*, could be combined with any of the possibilities on the dimension of interference aggregations, *the target of the interface*. At least six groupings of processes (object 7* to 12*) multiplied by at least five higher levels of aggregations of interferences (target f* to j*) already results in a total of thirty possible interfaces. The practicability and relative attractiveness could be investigated for each one. A more strategic approach may sooner give results. First, there is a search for a higher level of aggregation than found in current instruments. In that case a higher level of aggregation should be reached on at least one of the two dimensions. The main instrument of actual environmental policy, all over the world, is the installation permit. In their most aggregated form, permits do not exceed the level of "all processes on one location" covering "all relevant substances in a parallel way", that is the combination 3*-b*. This applies only to the most advanced permit systems, such as the still experimental integrated environmental permit in the Netherlands. With this, all environmental emissions and risks, related to all media, are treated together in one permit procedure. That permit preferably does not specify technologies, as is usually the case with permits, but only specifies allowable emission levels, as the target of this instrument (in Dutch: "doelvergunning"). It covers coherent groups of installations on one location as one unit. The analysis in the permit procedure does not take into account all contributions to all environmental problems in a formalized manner[2]. It is not yet a problem oriented approach. Increased aggregation at this interface should thus exceed that level, to pave the way for more macro instruments.

Secondly, one may choose a substantial increase on one of the dimensions and adapt the other at the highest level compatible with it. The compatibility refers to practical applicability. It does not seem possible yet, for example, to cover the life cycle per ECU consumed in different ways, combined with all substances aggregated according to their contribution to all problems, that is combination 11*-h*. In due course, however, this might become a practical possibility. One may first select the most relevant higher levels of aggregation of economic processes and combine these with the highest environmental aggregation practically compatible. To complete the search systematically, the highest

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1 Personal communication Shell Netherlands.

2 Different governmental implementing units remain responsible for different aspects of the permit. The unity, for the time being, is mainly a procedural one.

level of aggregation on the interference/environmental problem dimension as the target, will be combined with a low level of process integration as the object.

What are the main options for more aggregate process groupings? Ownership (4*) might play some role, e.g. as in liability and in environmental audits. However, there are risks involved in taking ownership structure as the object of a policy instrument. Avoidance of the object, one effective option for reduced environmental interferences if a material process is involved, may here lead to pathological results. Ownership-based instruments could, as a side effect, induce costly shifts in ownership structure that do not contribute to any environmental improvement at all. Extended liability may serve as one example. Its effects can not only relate to real prevention, as it undoubtedly does, but also to shifts on paper, as changes in the symbolic parts of the material processes concerned. In the US the publication of the start of a hazardous waste law suit may reduce the market value of a firm by several percent within days, negatively affecting all operations of the entire firm, see Muoghalu et al. (1990). If liability claims could take a substantial portion of profits, it would be advantageous to split firms in order to restrict both the effect of liability payments and their amounts. The hidden costs of this liability evading behaviour may be large. They result from the dynamic inefficiency caused by a distorted organizational structure. The net environmental effects are influenced negatively by this indirect effect route. With environmental audits (the version describing the direct effects of firms on the environment, not the procedural version, see the next chapter), similar evasive behaviour may result. If environmental audits on firms were to play a significant role, firms might "externalize" the parts of their operations that press hard on their environmental balance sheet, again with hidden costs, and also undermining the policy instrument. Taking into account these possible adverse effects, these two options are discussed further in the next chapter.

Sectors (5*) are groups of firms similar in some technical sense, as some aspect of their products, e.g. the food industry. As an object of policy they have the clear disadvantage that they constitute a statistical aggregate without any collective responsibility and that boundaries cannot be defined unequivocally.
Regions (6*) are the sum total of all economic activities in a geographical unit. Like sectors, they form a statistical aggregate of decision-makers.
They are not governed by a single independent-decision making unit, nor are they the subject of independent decisions of some non-governmental actor. With sectors and regions a boost in macro-policies can hardly be expected[1].

Nearly all other possibilities for aggregating processes, 7* to 12*, depend on the life cycle analysis of products (9*), as will be explained.
The unit of output of a process (7*) stays at the traditional installation level but takes the step towards an explicit trade-off with some symbolic economic value, here a product, as all higher levels of aggregation do. An example is the emissions per kWh of electricity. For a broad application, the unit of output approach has to deal with problems of combined outputs, e.g. of heat and electricity. That attribution problem is the same as the

[1] The sectoral option also would not be desirable for other reasons, as the sectoral policies developed seem intricately connected to a corporatist approach.

one encountered in the life cycle analysis. The emissions might also be displaced to other installations. Sulfur emissions of coal burning for electricity production, for example, may be reduced by pre-treatment of the coal in installations elsewhere, leading to emissions there. To prevent this shifting of problems to other processes at other locations, the total effects in the production column should thus somehow be taken into account. The production column, however, is not usually merely a single line to the origin. Most processes require different inputs from different other processes in the column to produce different products that are not all used in the production column for the product investigated. The analysis and the problems encountered are the same again as those occurring in life cycle analysis. If they are solved, a further step in the aggregation level has been taken. The grouping is then that of the production column of the product, option 8*. This option has been researched in environmental material analysis. With it, the environmental effects of different ways of making the same material may be investigated as well as the effects of using different materials with the same function. In that case, however, there will also be differences further in the life cycle, in use and in processing after the product has been discarded. It appears that a systematic treatment of these further processes, e.g. waste handling, requires the specification of the functional use of the products investigated. It is then no longer a production column but a production-consumption-waste handling column, that is the life cycle analysis of a product, option 9*, or even a group of products, option 10*. The life cycle analysis is now emerging, see Part Four.

Options 10*, 11* and 12* are all further aggregations of the life cycle of one product. The life cycle of consumptive directions (10*) may guide the development of our consumptive culture. The life cycle analysis of one ECU spent in different ways (11*) may similarly induce citizens to spend their income in an environmentally more responsible way. In this variant a broader indirect effect is taken into account than with the life cycle analysis of products, i.e. that consumers will go on spending their incomes. These two options are fully based on the life cycle analysis of products, and take it one step further. However interesting they may look, they may become operational only if that analysis has been established. The life cycle of all final products together (12*) does not yet open up vistas to interesting instruments. The conclusion here is that the life cycle, option 9*, is now the first major option for the aggregation of processes.

All material processes together, option 13*, is the highest level of aggregation possible. It is not a unit with some individual private actor in charge. Each one of them is addressed here. It may be effective in several combinations with interferences and their aggregations.

This first selection on the process dimension resulted in two main options, the life cycle of products, 9*, and the sum total of all processes, 13*. How might these two be combined with the options on the target dimension of this interface? The two main process options selected will be treated in turn.

The life cycle of products may be combined, technically, with any of the target options. The life cycle analysis required is developing as yet independent from policy instruments. See Guinée et al. (1992) for an analytic survey and Assies (1992c) for an historical survey. The simplest type of life cycle analysis restricts itself to one type of interference, option a*. The energy analysis developed in the Seventies can be seen as an example. A slightly more complicated alternative relates to groups of physically similar substances, c*. The arguments against the use of PVC often relate to such a relatively simple life cycle analysis, chlorine compounds emitted in production and waste handling being the only environmental interferences considered. In the LCAs conducted in the USA, it is usual to include all environmentally relevant interferences in parallel, that is option b*. The usually very large number of interferences causes serious problems in the environmental comparison of products. Hocking (1991) solved that problem by adding emissions by mass. A better approach seems the one common in Europe, to transform the large number of interferences into a limited number of environmental variables, in the form of problems. Currently, methods for thus translating interferences are operational only for certain environmental problems, see Heijungs et al. (1992) and are being developed for others, see Guinée and Heijungs (1993b). In the current situation the parallel analysis of all substances according to a number of environmental problems is possible, that is somewhere between option g* and option h*. With some effort, it should be possible to cover all main problems in a non-site-specific manner, realizing option h*. This approach results in the Problem Oriented Life Cycle Analysis[1], that is combination of object 9* with target h*. Some contributions to it will be made in Part Four.

If this combination has been developed practically, further extensions along both dimensions become feasible. At the object end the steps towards the life cycle analysis of groups of products and according to ECU consumed may be taken, to result in the combinations 10*-h* and 11*-h*. At the target end a certain preferably supranational body would have to take the daring step to specify the relative gravity of the environmental problems considered, resulting in option j*. Only this latter option will generally result in a clear statement on which of two alternative products are environmentally preferable, which groups of products are more attractive, and which of several alternative ways of spending an ECU is most attractive environmentally.

The second potentially attractive grouping of processes to be investigated is the sum total of all processes. It constitutes the total material activity of society. That object option might again be combined with any of the interference options. Practically, however, the analysis required is now mainly restricted to one individual substance, combination 13*-a*. It is the Substance Flow Analysis (SFA) now being developed in several countries. Technically, the extension of the analysis to groups of substances related to one problem, combination 13*-f*, is quite easy. An example is Kleijn (1993). The further extension towards the analysis of all substances related to one problem, combination 13*-g* also does not involve much more than the flows of the basic substances related to the problem, and their transformation into the problem contribution. A partial example on global warming is given in Part Five in the case on energy and global warming. As long as the number of substances related to a problem is small, the analysis required is manageable. This is the case for problems such as global warming, ozone layer depletion, acidification and eutrophication. For resource depletion and health problems the number of relevant substances is virtually unlimited. The analysis is then not practically manageable. The highest aggregation level now possible is that of groups of substances related to a

[1] POLCA would be a more precise acronym.

problem, that is combination 13*-f*. Some substances have a very short life span in the environment, e.g. VCM (Vinyl Chloride Monomer, a highly toxic and carcinogenic substance, the starting material for PVC). Because of this short life span, their effects are only local. If these were excluded from the analysis here, the relevant number of substances would drop substantially. The consequences of this omission for problem oriented policies cannot yet be surveyed.

Moving to the highest level of interference aggregation possible, the combination of object 13* with target h*, is the ultimate in aggregation for this interface. It implies the integrated analysis and aggregation of all interferences from all processes. It would allow the specification of the effects of increasing, or diminishing, the total production / consumption volume. The undifferentiated curbing of general economic growth is the only mechanism relevant, as long as there is technological progress[1]. The environmentally beneficial effects of limiting growth are undeniable in the long term. Limiting growth[2], now opposed generally for a variety of reasons, could then become feasible politically. For primarily practical reasons the relevant implementation mechanisms of this interesting interface will not be investigated further.

TABLE 2.3.3 INTERESTING MACRO COMBINATIONS OF GROUPS OF PROCESSES AS OBJECTS AND GROUPS OF INTERFERENCES (HERE: SUBSTANCES) AS TARGETS AT THE SOCIETY-ENVIRONMENT INTERFACE.

Object		*Target*	
One basic installation or product	1*►	◄a*	One basic substance
Connected installations	2*►	◄b*	All substances parallel
Processes on one location	3*►	◄c*	Group of similar substances
Processes of one owner	4*►	◄d*	One substance for one problem
Processes in one sector	5*►	◄e*	One substance for all problems
Processes in one region	6*►	◄f*	One group of substances for one problem
Processes for one output	7*►		
Processes in production column	8*►	◄g*	All substances for one problem
Processes in product life cycle	9*►	◄h*	All substances for all problems
Life cycle / groups of products	10*►	◄i*	One substance for total environment
Processes in life cycle / ECU	11*►		
Life cycles of all products	12*►	◄j*	All substances for total environment
All processes	13*►		

[1] In times when the dominant idea is that economic growth is a prerequisite for increased effectiveness of environmental policy, as stated in the Brundtland report, growth reduction hardly seems a sensible option. There now seems to be a strange emphasis on extra production with both environmentalists and brute economists, at times when both the absolute production and the production per head are at an all time high. The German The Ministry of Economic Affairs urged industry (June 1992) to start working on Sundays, in order to increase sales in internationally competitive markets. Nearly the whole political spectrum in the Netherlands agrees that labour participation in paid work has to go up. The reasons are budgetary, to reduce the payment of social security benefits, cultural, to integrate ethnic minority populations better in society, and emancipatory, to make women more equal to men in the external labour respect. For environmental reasons, however, a lower production - through fewer people, working fewer hours - is clearly to be preferred. A basic income to everybody could reasonably accommodate the cultural, emancipatory and environmental aims. See v.d. Veen 1991.

[2] The object implies the **non**-selective limitation of growth. Other instruments, like taxes on all processes for all emissions of substances, combination 13*-b*, may be have a selective limit to growth as an effect.

One combination of options, so far ignored, might be interesting. It is the high level of integration of interferences, say all substances according to their contribution to the total environmental problem, j*, with the grouping of technically connected processes, 2*, as is now common for permits. This combination, 2*-j*, would seem to allow the overall environmental optimization of processes. However, this is only true if no concomitant effects for other processes would result. If these effects are also taken into account, the analysis would allow an overall environmental assessment of processes. In such a comparison, the criterion for comparison, the factor which is the same in both processes could scarcely be anything other than the function of the process, which is to say its products or services. With these additions it is in fact the life cycle analysis of products that emerges again. The latter analysis refers to all processes required for the functioning of a product. The former refers to the effects through all processes required for the functioning of an installation. The combination, thus enlarged, of the functioning of one process in terms of all substances according to their contribution to the total environmental problem, combination 1*-j*, will not be worked out separately. It would be a major improvement on current practice if general rules on installations, for example, could be specified in such terms for all non-local effects.

2.3.6 Other targets and the efficiency of targets

Other targets?

The definition of object and targets given in the preceding section excludes some instruments that are customarily seen as instruments for environmental policy. Excluded, for example, are design standards that lack an environmental target; these are not specifically related to interferences or the environmental problems resulting. Another example of an excluded instrument is a fixed percentage investment subsidy for technologies that are supposed to be relatively environmentally beneficial, since these subsidies are not related to the interferences the subsidized process causes. The limited range of "pure" instruments makes it impossible to place those defined differently into a comparative perspective. Some of the extra targets encountered will be tentatively defined.

Certain instruments have a target that goes further in the effect chain considered appropriate here for material interferences with the environment. These generally relate to disturbances, a type of interference that is disregarded here. An example is the prohibition of trade in nearly extinct species, dead or alive, to prevent them being captured. Also, some effects of substances much further along the effect chain might be quantified in monetary terms. The liability instrument is an example, as applied, for example to the damages of oil spills in offshore exploration and exploitation in the USA. Thus other environmental targets possible are

◇ environmental damages.

As indicated above, the quantification in monetary terms of most environmental effects is simply not possible. Liability will thus remain limited in its application.

Some instruments have a target that does not go far enough along the effect chain; they have a target whose interferences are not even identified. Such instruments have the longest history in environmental policy, going back for centuries. Technical standards for processes and products, and for behaviour may all be formulated without any explicit reference to interferences and their effects. The assumption, concealed at the level of the instrument, is that through these technical or behavioral standards the environmental interferences will diminish or that their composition will become more beneficial to the environment. Such targets may be near interferences, or farther away. They thus may function as a proxy for real environmental targets. Major options for such almost environmental targets are

◇ amounts of a potential hazardous substance or depletable material in a product or installation, that is combined with object 7*

 (cadmium-free pigments and stabilizers; carbon tax on fuels; PVC in packaging materials aimed at preventing the emissions related)

◇ technical or behavioral prescriptions of a product or installation, again combined with object 7*, or perhaps with a partial life cycle, object 9*

 (speed limits for cars aimed at limiting NO_x emissions)

◇ design of a material, product or installation, again combined with object 7*, or, if the life cycle is taken into account, object 9*

 (design standard in permits stating best available technology, BAT; for products analogous, as in minimal electric efficiency of household appliances)

◇ type of disposal of a product or installation possible, with only the partial life cycle as an object, option 9*

 (recyclable cars and recyclable packaging materials)

They would become interesting again from a theoretical point of view if the "pure" instruments for environmental policy developed here were not practicable or not effective enough. In this study they are mainly left out of consideration.

The preliminary choice on object and target should take into account the criteria with which instruments and policies may be evaluated. Efficiency and equity are two main criteria. Efficiency is based on a minimum of side effects, through, for example, problem shifting. Equity, e.g., requires equal treatment of equal cases. The choice of target determines the possible efficiency of environmental policy to a large extent. The nearer the target is to the final values involved, the fewer side effects will occur and the more efficient an instrument can function. See Nichols (1984, e.g. p.160) who strongly defends this position on efficiency grounds. Going to the highest possible level of a target thus increases efficiency, in principle, since a minimum of side effects will occur. Describing instruments in terms of "energy", "waste volume" or "recycling" is thus in principle inferior in terms of efficiency compared to environmental interferences as a target. The problem target again improves on the material interference target. The economic value of effects, as the target in liability rules, is superior again in preventing costly side effects. The second criterion, that of equal treatment, requires that policies apply equally to all cases. In that case, economic value is a less attractive target since its possible application is only very limited. The more aggregate the object of the instrument, the more probable an encompassing application becomes. Thus, the provisional choice of two main options of aggregated interfaces not only is as macro as seems reasonably possible but, *a prima vista*, is also in line with further evaluative criteria for instruments.

2.3.7 Conclusions

A number of types of interfaces at the boundary between society and environment can be specified in terms of objects and targets. More macro society-environment interfaces may be generated by means of these two dimensions. One is process related and specifies

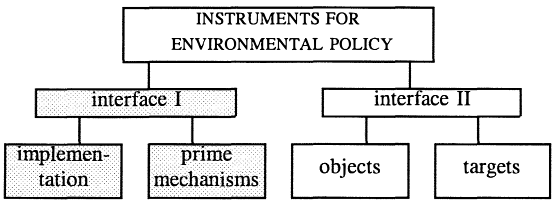
several groupings of processes, as the objects of instruments. At this process dimension, the most promising objects for the development of macro instruments are the life cycle of a product, option 9*, and the sum total of all processes, option 13*.
The other dimension, that of the target, aggregates interferences, through physical aggregation and through their translation into problems. The problem related target aggregation, option d* and higher, seems the most relevant tool for more aggregated targets.

The life cycle is best combined with all the substances related to all supra-local, non-site-specific problems, that is the combination of object 9* with target h*, see table 2.3.3. The current developments in Life Cycle Analysis (LCA), an analytic tool not yet applied in instruments, proceed in that direction, at least in Europe. If that interface is established, it seems a small step to indicate the relative importance of the problems concerned and take the further step to a total environmental assessment, that is option 9*-j*. If this is well established, further aggregations to the life cycle of different consumptive directions and the life cycle of different ways to spend an ECU, might follow quite easily. These latter two options allow the definition of a "life style" that is relatively environmentally friendly.

The sum total of all processes may first be combined with some basic substance. Then the Substance Flow Analysis emerges, that is the combination of object 13* with target a*, or c*. This analysis may quite easily be aggregated to groups of substances related to one problem, combination 13*-f*. The step to all substances related to one problem, combination 13*-g*, is also easy for some environmental problems though not all.

These two main combinations seem to form the basis for the most macro, efficient and equitable instruments that can now be designed. They will be worked out in Part Four as a part of complete instruments, i.e. in relation to their most relevant working mechanisms for implementation.

2.4 THE SYMBOLIC INTERFACE BETWEEN GOVERNMENT AND SOCIETY: INSTRUMENT MECHANISMS



2.4.1 Introduction

The two interfaces, one between government and society, the other between society and environment, together define an instrument for environmental policy, in the strict sense. The primal working mechanism has been defined as the main element in the interface between government and society. Thus adding a working mechanism to the two main macro-interfaces defined in the preceding chapter is all that remains to be done in this chapter.

The problem-oriented life cycle analysis may be combined, for example, with a prohibiting mechanism. An allowable maximum contribution to each of the problems concerned could be formulated, e.g. standards for some class of products. Any product exceeding one or more of the limits as formulated in these standards would simply be forbidden. A financial instrument mechanism would be a tax on each problem, proportional to the contribution to them by a product in its life cycle. A cultural instrument mechanism could be the publication in a "black list" of all products exceeding at least one standard for allowable contributions to each problem. A structural instrument mechanism could be the burden of proof reversal in liability suits for all products exceeding one or more standards, with the creation of a monitoring organization that establishes the facts on every product.

Unfortunately, the situation is not so simple, for a number of reasons. First, there are different ways each main mechanism might be filled in, e.g. at a more, or less, macro level. The financial instrument, for example, might also be formulated as a tax per unit of problem contribution, at a level to be specified for each problem. That implies a much higher level of aggregation. The same taxes then apply to all products, not just to specific classes of products. Thus a design job, directed at filling in each type of working mechanism in the most macro manner, is still wanting. Surveying instruments from the literature can be a good "bottom up" help in this design procedure.

Secondly, the main mechanisms have been formulated in a highly abstract manner, e.g. "structural mechanisms", "cultural mechanisms", etc. This hardly indicates a specific course for an administrator to follow. Working out the main mechanisms in more detail would give more "top down" guidance in the design procedure. This will be done, first, on the basis of the work of two Dutch economists, both of whom have been engaged in practical policies, macro-economist Tinbergen and institutional economist Zijlstra.

Thirdly, the ideal types to be developed might not be applicable in all situations or might not be effective enough environmentally. So a range of instruments broader than that of the ideal ones might be required for the development of reasonable policies, to fill the smaller or larger gaps left by the ideal ones. A survey from the literature could indicate candidates for this, minor or extensive, supporting role.

Fourthly, there is a vast body of literature on policy instruments in general and on instruments for environmental policy in particular that reflects the thinking on the subject. A survey of relevant literature from several disciplines related to the subject of policy instruments, not necessarily complete but covering main approaches, will help to clarify the framework and also show in which sense the framework and instrument classification developed here differ from these other approaches.

The most concrete *aim* of this chapter is to produce an extensive list of descriptions of the most macro instruments of each type that might be needed for a "complete" macro environmental policy. A preferential classification of these instruments is not yet given, apart from incidental references. That normative analysis is the subject of Part Three.

Before the organisation of this chapter is described, there first are a few words of caution mainly stating what will not be done. The term "instruments" may denote anything that is instrumental to some aims, here environmental aims. In that sense demonstrations and actions, as by Greenpeace, as a means to influence public opinion, are an instrument that may regulate private behaviour and through it the environment. Such internal workings of society can be very important but are not the subject of the instrument discussion here. Similarly, the internal workings of government, very relevant of course in policy making, are not at stake here in the discussion on instrument mechanisms. Thus, governments may issue laws, or rules with a similar status, that state environmental aims like ambient standards, they may decide on the means to be applied by other governmental bodies, they may fix procedures to arrive at practical policies, and they may give rules for implementation. EC directives, for example, bind national governments to certain decisions, that in turn bind local governments and agencies, that finally change the behaviour of these regulating bodies towards regulatees, that is the parties regulated. All these items may legitimately be termed "instruments of environmental policy". Again, the instruments internal to the government system, that is the instruments to regulate the regulators, are not the subject of the instrument discussion here. Here policy instruments only denote the mechanisms used by government to influence the non-governmental part of society.

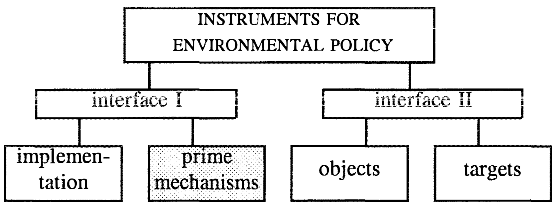
As may be clear by now, the situation at this interface is different from that of the other. There, the most basic unit could be defined quite concretely, as one process with one interference. For this interface no such precisely defined basic unit can be formulated. The basic items elements that flow between government and society are not as easy to define as the material flows between society and environment. The flows here are not material but symbolic. Essentially it is the very diverse means of power that are involved. Money, as a government-backed claim on future goods and services; threats and promises of negative and positive sanctions of diverse types, including monetary ones; sanctions; information on what is good, bad and possible; authority and legitimacy; procedural rules on how private individuals and organizations should act; and well as procedural pledges on how government will react to certain types of private actions. Such diverse mechanisms can hardly be reduced to a single common denominator that still has a concrete meaning similar to "one material economic process with one interference" as at interface II.

The main idea remains the same however. This interface I links government to society, through defining the working mechanism or operation of instruments. These constitute the link that translates governmental processes into societal ones. As processes of implementation on the government side are quite malleable, the main characteristic to start the definition with is the prime mechanism of change in society. That mechanisms is set into motion by addressing the instrument to some responsible entity, such as a firm, groups of firms or all firms, or one person, a group of persons or all individuals.

There is always a relation between addressees and the object of the instrument; the addressees are those responsible, directly or indirectly, for one process or groups of processes as specified in the object of the instrument. There may be a direct relation. With individual permits, for example, the addressees are the owners of the process or process concerned (or their representative). The relation may also be more complicated, as with ecolabelling based, for example on problem oriented life cycle analysis. There, the addressee is a person in his capacity as consumer. He makes a choice between products - his responsibility - while most of the processes concerned are the responsibility of various producers and waste handlers. When establishing an authoritative system of life cycle analysis, as one possible cultural instrument, the link between addressee and object becomes still more indirect. The system's addressees are all persons and organizations in society. The objects are the life cycles of all existing or considered products.

The following section, 2.4.2, first deals with classifications and typologies as defined by Tinbergen and Zijlstra, and others, of the instruments of economic policy. Economic policy has long been using mainly meso and macro instruments, here together called "macro". Their distinctions will be fit into the main scheme used here, if possible. Application of these approaches to policy instruments might at least bring more real content into the now still quite abstract categories of prohibitive, financial, cultural and structural prime working mechanisms in society that instruments may have. Next, the instruments as seen in several disciplines are surveyed, in section 2.4.3. The aim there is not to give all instruments encountered a place in a theoretical superstructure of instruments. It is to determine if there are elements and ideas that may help develop macro instruments for environmental policy, or to show, by contrast, how other approaches differ from the macro oriented one developed here. The main emphasis is on those instruments distinguished in the economic literature. The several instruments encountered are specified as to their working mechanism, as a main part of the one interface, and as to their subject and target, the elements of the second interface. They then may be ranked according to the dimension of "macroness". The scope of instruments investigated is extended to those that are environmentally relevant, albeit not strictly defined in terms of the society-environment interface as developed here. Then, in section 2.4.4, the results will be put together, with a specification of the most aggregate instruments for each of the four main types of societal working mechanism. It forms the first main result in the design procedure for macro-environmental policy. Section 2.4.5 treats the governmental side of the interfaces, primarily related to the implementation required. This will be done in terms of strategic requirements, such as style of regulation and types and amounts of manpower. Specific actions required for each instrument will not be treated systematically. The chapter ends with conclusions, section 2.4.6.

2.4.2 Macro instruments for economic policy classified



This section begins with observations on the instrument classifications of theorists on macro-economic policy Tinbergen and Zijlstra, and on later changes to Zijlstra's scheme by Geelhoed and by Hellingman and Mortelmans. The result is a second version of the main classification of instruments, differentiating somewhat the four main types of the preceding chapter. The analysis concerns prime working mechanisms mainly, at the interface I between government and society. Some possible differentiations as to level of aggregation at interface II, that between society and environment, are indicated. When reviewing other disciplines on instruments, in the next section, the emphasis is similarly on working mechanisms.

Tinbergen makes a very formal distinction between three levels of instruments. There is quantitative policy, qualitative policy, i.e. changes in structure, and most fundamentally, reforms, i.e. changes in foundations. Examples of quantitative instruments are the amount of money circulating, the interest rate and the level of taxes. Such changes may be brought about at short notice. Examples of qualitative instruments are the introduction of competition in a formerly monopolistic industry, the rationing of some products in certain situations and fixing prices to correct for inadequate market mechanisms. Non-environmental examples of the latter two are in EC agricultural policy. Such changes are less frequent and work on a longer term. Examples of changes in foundation are the introduction of a system of minimum wages; of fixed instead of fluctuating exchange rates, of industrial democracy, and of improved access to education. Such changes are being brought about even less frequently. Their importance is usually only in the long run. After the introduction of qualitative and fundamental changes, the working of the system is changed as seen in the more short term models for quantitative policies. With qualitative reforms and changes in structure, new instruments may become available that require quantitative policies, as in the example of agricultural policies, or they may not, as with some changes in liability rules.

Tinbergen (1967) sees no indication of how, at these higher levels, instruments may be developed and applied systematically.

> *"In the case of qualitative policy even the enumeration of all alternatives would be hardly possible according to any system; and a choice still less therefore." (p.150). "The same is true, even to a larger extent, for the more fundamental changes in social organization called reforms; .."* (p.153).

Making a modest contribution to such development in the area of macro instruments for environmental policy is thus the not-so-modest aim of this study.

In terms of Tinbergen, the design and introduction of new instruments is one of qualitative and fundamental change. Only their implementation may require quantitative choices but with many instruments that is not necessary. Introducing rules on industrial democracy is an example. Tinbergen's subject was macro-economic policy. The qualitative and fundamental changes he considers thus also belong to the realm of macro-policies. In general, such structural changes do relate to highly aggregated aspects of society, in the more loose terminology used here, to the macro-sphere. Tinbergen, a social-democrat, was developing practical macro-environmental policy as a means for socialist (socio-democratic, non-communist) control of society, especially in its economic aspects. There was not much doubt about the aims nor was there much doubt about the possibilities to reach them. The Keynesian revolution in macro-economic theory, combined with the smooth introduction of qualitative changes and reforms made the world malleable.

In environmental policy, similarly daring reforms and qualitative changes are required. Their introduction would, in Tinbergen's line of reasoning, always constitute a structural change, in his terms a "fundamental change in social organization". The actual administration of the new instrument would be quantitative, at least for some of the changes introduced. In the context of environmental policy, the adjustment of the level of a "problem-contribution-tax" would be quantitative policy, as would the adjustment of the allowable "problem-weighted heavy metal emission level" in a permit. It seems preferable not to name instruments after the changes required for their *introduction,* but only after the mechanisms they exhibit when functioning. Reforms are thus structural only if their effects work at the structural level. Reforms would be financial instruments, e.g., if their functioning requires quantitative changes in prices. Introducing new prohibiting instruments, like problem related design standards, would also be a reform. The adjustment of standardized amounts would constitute quantitative policy. Although the general spirit of Tinbergen's model-based approach has inspired the approach developed in this study, the specific contribution to main categories of general instrument theory is limited.

Zijlstra, another Dutch economist, but a Calvinist "anti-revolutionary" who later belonged to the Dutch Christian Democratic Party, took another vantage point. He was interested more in the relation between government and private society as a subject in itself. That subject concerns, in our more contemporary terms, the government-society interface. In the discussions on the tasks of the newly developing European Economic Community, he wrote a document on policy instruments and the very much related economic order desired. The policy domain was again economic, concerning economic policy and competition in the EC and its member states, see Zijlstra (1966, with Goudzwaard). Zijlstra, after surveying the instruments of economic policy practically available, set out to clarify the types of instruments available, a subject on which there was "a lot of unnecessary confusion" (p.32). He distinguishes six main types, with some additional variants (pp.32-3). They are described in table 2.4.1 below[1]. Geelhoed, a more liberal, "sadder-but-wiser" social-democrat, takes the Zijlstra types as a starting point for an instrument analysis in a large project aimed at restructuring governmental organization in the Netherlands. He makes some changes and additions, see Geelhoed (1983). The types of Zijlstra and Geelhoed are surveyed in table 2.4.1.

TABLE 2.4.1 TYPES OF POLICY INSTRUMENTS AS DISTINGUISHED BY ZIJLSTRA (1965) AND, RELATEDLY, GEELHOED (1983)

Zijlstra[2]	**Geelhoed**
Type 1 Information and prognostications No aims or tasks are mentioned. Subjects change their behaviour freely, adjusting to the situations as the information shows, if they want so.	*Type 1 Information and prognostications*
Type 2 Tasks set by government for itself Private subjects remain absolutely free. They can adjust their behaviour according to their expectations of what (non-regulatory) governmental bodies will do.	*Type 2 State procured facilities that shape the decisions of citizens*
Type 3 Means for macro aims brought about by individual changes in behaviour not themselves influenced directly* Private subjects are influenced through a change in the situation. Their own motives and free choice remain unaffected. Regulating the amount of money in circulation influences the prices on virtually all markets, e.g. attempting to curb the level of inflation.	*Type 3 Interventions influencing citizens in a general way* *3A Quantitative macro-interventions* *3B Normative general interventions upheld by private law* *3C Normative general interventions upheld by criminal law*
Type 4 Direct influences in terms of specific support and facilities, without coercion, and without public and private interests being geared[3]	*Type 4 Unconditional taxes and unconditional outputs*
Type 5 Direct influences on private organizations gearing them to publicly decided on interests, without coercion* There is a mutual relation here between regulator and regulatee each influencing the other.	*Type 5 Conditional taxes and conditional outputs* *5A Unilaterally applied subsidies and taxes* *5B Consensually applied subsidies and taxes*
Type 6 Specific coercion of firms to behave according to the aims set by government*	*Type 6 Unilateral coercive interventions*
* A possible further subdivision as to sector is indicated by Zijlstra for types 3, 5 and 6.	*Type 7 The exclusive competence of the state to engage in certain activities*

[1] The terminology of my translation here from the Dutch abstracts somewhat from the context of economic policy he had in mind.

[2] Zijlstra, contrary to Geelhoed, does not give names to the types, only a number with description and examples.

[3] Hellingman and Mortelmans 1989, p.204, give Zijlstra's list in a slightly different interpretation. Type 4, for example, is described as *unconditional levies and unconditional outputs*. Zijlstra does not specify the levies in his type 4, however, nor in any other of the types. Their list seems to be based on the types of Geelhoed 1983. He is explicitly inspired by Zijlstra but follows him only loosely.

Koopmans (1942), a forerunner in the same tradition, uses similar instrument types to typify states according to the ways their governments predominantly control the non-governmental part of society. That may remind us of the wider implications for society of the choice of policy instruments.

The attractiveness of the Zijlstra typology is that there is one principle for ordering the types, i.e. according to the amount of freedom taken from, or complementarily left to, the economic subjects. Geelhoed's changes and additions do not fit into this pattern. Zijlstra's most freedom leaving, least constricting, types, 1 and 2, are based on the communication of neutral information (type 1) and a limited type of normative information, and only on the aims set by government for itself (type 2). If this second type is broadened to normative information in general which also concerns society, the types fit into my scheme as two cultural instrument types. They both require a cultural model of society to assess their functioning. The information may be of a general type, e.g. concerning the way environmental effects of processes are to be assessed. Or it may be of a specific, less aggregated type, e.g. exhorting people not to use plastic bags. Geelhoed has left out the normative type of information altogether, or automatically includes it under "information".

Geelhoed's type 2 is altogether different; it is state procured facilities, to be understood as material facilities, that shape the decisions of citizens. An example is the public containers for the collection of used bottles. Although a relevant category, I prefer not to treat this instrument type separately here, for a systematic and a practical reason. First, in the model system chosen, government activities do not have a material component. All material processes reside in society, regulated by government only by symbolic means. Of course, governments may induce or force organizations in society, collective or private ones, to supply certain facilities. At the instrument level, however, no differentiation between public and private organizations is made in this respect. For the actual functioning of instruments, as modelled independently of the instruments, it might of course make a difference whether public or private organizations are regulated. This is a difference similar to that between large and small private firms, which is not usually incorporated into instruments either.
Secondly, there are practical reasons to exclude state procured facilities as a separate instrument type. It is often difficult to decide if some installation owned by a corporate body is public or private. Governments may own some of the shares, as with many water purification facilities; they may have statutory seats in boards; private sector firms and persons may unite with some recognition of a special status by state organs, as in the former German 'Genossenschaften' (see Klavorick et al. 1973); and regulations on some purely private firms may be so strict as to make them actually behave in accordance with governmental aims on a day-to-day basis, as with some monopolistic utilities. Or, state owned facilities might be totally "unregulated", coordination being left to the markets for labour and commodities, combined for example with a state monopoly on the activity. In that case apparent state production of the product is fully equivalent to the functioning of private firms in a monopolistic market[1].

[1] This may partly explain the severe damage to the Eastern European environment. Corrections on market incentives, instruments of type 4 and 5, might easily have been used by communist states.

In the example of supplying public glass-collecting containers (anyway not an instrument for environmental policy in the strict sense used here as an interface with the environment is lacking) several types of implementation interface are possible. Such a governmental provision of material facilities might be covered by Geelhoed's type 2, as discussed here. It might be combined with type 7, i.e. the exclusive competence of the state to engage in that activity, or might even be covered by it. Or it might be covered by Zijlstra's type 6, where governments may fully decide how facilities should operate, be they owned by them or not. Or governments may pay a firm for the contract to supply the service, thus covering the costs of an otherwise financially unattractive activity, i.e. type 5B of Geelhoed, a category lacking with Zijlstra. Or governments may subsidize any amount of glass thus collected, type 4 or 5A of Geelhoed, as is usual with the collection of paper from households in several countries and cities in Western Europe. Or governments may put moral pressure on the producing firms involved to supply the collection of their waste packaging, covered perhaps by Geelhoed's type 1, as seems to have been the case with aluminium cans in the United States. Thus, it seems advisable for these reasons as well to leave procurement of (physical) facilities out of the typology of policy instruments.

Zijlstra's type 3 are the macro-instruments, that only indirectly influence relevant behaviour. This separate type seems to introduce a dimension that is relevant for all the other types, i.e. the level of aggregation of instruments. I prefer to treat that dimension separately, indicating possibilities for aggregation within and between each of the four basic types. As indicated in the discussion on Tinbergen, macro instruments would either require quantitative implementation, e.g. through financial, prohibitive and cultural instruments, or they could be structural instruments. Geelhoed distinguishes three subtypes. Quantitative macro intervention (3A) is a speciation of Zijlstra's type 3, leaving out the qualitative macro intervention termed cultural and structural instruments by me. His types 3B and 3C, normative general intervention as upheld by private and public law respectively, brings specific juridical forms into the instrument classification which seems wiser not to do, see below. Normative intervention upheld by criminal law, his 3C, would probably be prohibiting instruments, of a general, macro, nature in the terminology used here. "Normative general intervention", if stripped from prohibiting and financial elements, then belong to the cultural instruments for environmental policy.

Zijlstra's type 4 may be interpreted as his first type of financial instrument. He states the subsidy only. Curiously, there is no explicit place for taxes in his scheme. He might have included general taxes under "means for macro aims", type 3. This would not seem a sensible option as general subsidies may influence behaviour in a very similar manner. Geelhoed's type 4 and Hellingman and Mortelmans' (1989) interpretation of Zijlstra's type 4, is a category of financial instruments with general applicability. Both name this type 4 "unconditional taxes/levies and unconditional government outputs". The latter category includes financial support, as some sort of subsidy. Material support in the form of public facilities is also included in Zijlstra's and Hellingman and Mortelmans' type 4, the same item as Geelhoed's state procured facilities, his type 2. As indicated there, the public versus private ownership aspect is not to be included in the instrument definition. The public procurement then is a special case of type 6. Type 4 is thus reduced to generally applicable financial instruments only.

Only Geelhoed divides type 6 into different subtypes (not in table 2.4.1), according to several criteria:

◇ how deeply the intervention influences private decisions
◇ how selective, or general rules are
◇ what the judicial status of the rules is and what the possibilities for official's discretion are
◇ whether the content of rules is primarily normative or primarily procedural
◇ the complexity of rules.

The most relevant distinction in terms of both the level of aggregation and the degree of freedom taken is the selectivity of rules. A crude dichotomy splits prohibiting instruments into those that are generally applicable to classes of cases, and those that apply to single cases. Prohibition here includes both the conditional (it is forbidden unless) and the unconditional types (it is forbidden to).

One further restriction in the approach to instrument design may be noted. The instruments are restricted to substantive, as opposed to procedural types. Their implementation of course requires a certain procedure. Their target, however is a normative substance of reality. An example of a procedural target of an instrument is that all investment decisions in larger firms may only be taken "after review by the responsible environmental manager of the firm". Some observations on the discretion of officials are made in section 2.3.6 on administrative implementation. Another example is the obligation to produce an environmental impact statement (EIS) before decisions on an investment project may be made.

A systemic addition Hellingman and Mortelmans make, referring to Eucken (1952, their reference), an institutional economist, is to place the instrument types of Zijlstra in a first shell, surrounded by other societal steering mechanisms in next shells. The Zijlstra types all belong to an what they call *output oriented* method of steering. The *input oriented* method, named "ordering policy" or "conditions creating policy" is placed parallel to the Zijlstra types (p.206). These policies create the structure within which processes can function. Examples are rules on market entry, as in cartel law. Both the input oriented and the output oriented type of steering use public law, or statutory law in Anglo-Saxon countries, as a basis.

A next shell distinguished by Hellingman and Mortelmans is that of private law, which regulates property rights, liability rules and contracts. It is an input oriented method of steering as well. The distinction between the judicial status of public law and private law, necessary for distinguishing the two shells, is based on specific systems of law in different countries[1]. There does not seem to be any point to including such distinctions in the status of rules in the interface. They would lead to "continental" and "Anglo-Saxon" typologies of instruments. Hence the distinction of public and private law is left out here and the two shells of Hellingman and Mortelmans reduce to one. They are part of the structural instruments that create and change the structural aspects of society. They

[1] The distinction between private and public law is typically based on the continental systems of law as introduced by Napoleon. Public law further subdivides into administrative law and criminal law. The main analogous Anglo-Saxon distinction is that between custom-based (common) law and statutory law. In statutory law rules on liability can be formulated that, on the continent would be part of private law. Communist countries, including China, and all formerly communist states, have introduced the continental system.

Type 5 again consists of financial instruments mainly, if public facilities are subtracted in the way indicated above. The difference from type 4 is that the conditional taxes and outputs are now more intricately connected to specific developments in firms and sectors. Geelhoed calls them "conditional taxes and conditional outputs". This seems quite in line with the intentions of Zijlstra and to fit in his scheme.

In the treatment of economic instruments, type 4 and 5, there are subtle differences between the authors mentioned. Geelhoed sees type 4 as consisting of general taxes and of general transfer payments from government to private persons and organizations, as in social security payments. Type 5 he sees first as general (5A) taxes and subsidies and, secondly, as case oriented (5B). Zijlstra and Hellingman and Mortelmans, on the other hand, see the latter difference as the distinguishing characteristic between type 4 and type 5. Geelhoed has to make a distinction between general unconditional taxes, his type 4, and conditional unilateral taxes, his type 5A. This seems difficult in practice, as unconditional taxes are also unilateral. In both cases certain quantifiable circumstances decide the level of the tax. Zijlstra and Hellingman et al. are followed here. The main difference between the two types is their level of aggregation. Unconditional, unilateral taxes and subsidies apply to cases defined generally. Conditional, consensual taxes and subsidies are case-specific. They both fall into my category of financial instruments, with a differentiation as to how macro the character of these economic instruments may be.

Zijlstra's "gearing private organizations to public interests, without coercion" and Geelhoed's "consensually applied" financial instruments might indicate a type of decision-making on instruments that is mixed with their implementation. The formal distinction adhered to between on the one hand, a decision and on the other, the implementation does not allow the introduction of such a procedural characteristic as part of the instrument. Of course, societies may differ profoundly as to the degree of consensus required for both public and private decisions. That, however, is something to incorporate in the analysis of the political decision-making process, not a subject of this study, and it is something to incorporate into models of society, that are not worked out in this study.

Zijlstra's last type, unilateral coercive intervention, type 6, covers the regulation of private activities through coercion. They cover all "prohibitions". As indicated above, they also may cover the regulation of government owned facilities, including the governmental provision of facilities, even if these facilities are the "exclusive competence of the state", i.e. Geelhoed's type 7. Geelhoed's distinction into types 6 and 7 is thus negated. In terms of regulation, the material production of governments is treated the same here as any other production and consumption process. Coercive intervention, direct physical regulations, command and control techniques, or whatever they may be termed, apply to such collective facilities in the same way as to private ones[1]. Their functioning of course may differ due to such ownership aspects as it also will because of other circumstances.

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[1] For the other types of instrument there is no differentiation in relation to public or private ownership either.

encompass the reforms, as changes in foundations, and the qualitative changes of Tinbergen, at least those that do not require a quantified implementation. Rules on the entry into an industry are an example. It is not at all clear, however, whether this addition really is necessary. The non-quantitative elements of type 3, both Zijlstra's "means for macro aims" and Geelhoed's "normative general interventions", could cover such structural instruments.

One of the main differences between Zijlstra and Geelhoed, and Hellingman and Mortelmans is that the former two place these structural instruments in the middle of their list, as type 3, indicating a higher level of freedom for their (differing and partly broader) types 1 and 2, which encompass the cultural instruments. I would prefer to follow Hellingman and Mortelmans in this respect and place the structural instruments on top, as the type that leaves the most freedom. Their application might also be the most general one, applying to all symbolic economic processes in society.

What has been learned from the discussion of these authors on economic policy? First, that the typologies of even very close followers of a main approach may diverge from the original in a fundamental way, even if seemingly small changes are introduced. The meaning of categories starts to change and the overall principle of Zijlstra (increasing level of freedom) is no longer valid. One cannot hope to find the "smallest common multiple", as a metasystem in which even very similar approaches to instrument classifications would fit together. This means that a somewhat postmodern situation has to be accepted in which, at best, approaches in one or more respects can be compared, without fully covering their combined content. When using approaches of others, one thus uses only single elements or parts and one fits these into one's own framework where this seems advantageous. This is what will be done here.

The second question thus follows, what can the approaches surveyed teach us about the relevant elements to incorporate into the as yet barren structure of the four main types of instrument mechanisms? It appears that they all distinguish between levels of instrument generality, *within* each of my main types. For each, a different level of aggregation or *macro-ness* is possible. A differentiation in this sense seems fundamental to them as well. Thus, the classification of main types of instruments as to level of aggregation, as presumed until now, must be qualified. The position taken here is that the macro character of instruments may vary within types to such an extent that no ordering of the main types in that respect is possible directly. However, the maximum level of generality possible seems to increase from prohibiting, through financial and cultural instruments, to structural instruments. The variation according to type is stated here, for the moment, as a dichotomy, see table 2.4.2, with some examples on instruments placed in brackets.

Thirdly, a more negative result is that a number of distinctions used by the authors surveyed till now do not seem basic. The juridical status of rules, introduced at different places by both Geelhoed and Hellingman and Mortelmans, varies extensively between countries and does not seem to be a vital characteristic of instruments. Nor does this seem to be the case with the procedural aspects introduced into some instruments.

TABLE 2.4.2 MAIN TYPES OF POLICY INSTRUMENTS ACCORDING TO THEIR PRIMAL WORKING MECHANISM, DIFFERENTIATED AS TO LEVEL OF AGGREGATION.

Primal working mechanism	Differentiations as to aggregation	
Structural instruments	◇ high	General normative arrangements (including rules on property, liability, contracts and public sanctions) and general institutional arrangements (including market structure and research institutions)
	◇ low	Idem, but case specific (e.g. regulations on the liability for solid waste)
Cultural instruments	◇ high	Factual and normative information of a general nature (relative importance of different environmental problems stated)
	◇ low	Factual and normative information of a specific nature (environmental attractiveness of different products stated)
Financial instruments	◇ high	Unilateral taxes and subsidies, generally applicable (emission tax)
	◇ low	Consensual taxes and subsidies, negotiated per case (investment subsidy)
Prohibiting instruments	◇ high	General prohibitions (design standards)
	◇ low	Case specific prohibitions (rules in individual permits)

2.4.3 Discipline-related policy instruments

There exists an extensive literature on policy instruments. All related disciplines have their own terminologies and distinctions. The distinctions made so far related to students of economic policy, with an emphasis on instruments with a higher level of aggregation than that of the individual firm. Regulating the economy, as a subject, seems closest to environmental regulation of what is mainly the same economy. Of course, the targets differ. A cursory, not-exhaustive survey may indicate how other disciplines might be related to the distinctions made here. The survey has a dual purpose. First, the classification scheme developed thus far may be compared to other ones, to indicate similarities and differences. Secondly, elements and parts of other categorizations my be added to the typology here. The disciplines commented upon are (1) political sciences, (2) public administration, (3) law and welfare theory, and (4) economics, concluding with (5) a final discussion on disciplines.

1. Political sciences and macro-sociologists

Political scientists and macro-sociologists speak of the means of power or the means of control. This would be very similar to the "prime working mechanism" as a basis for the instrument typology used here (for interface 1).

Etzioni (1968), for example, distinguishes the means to make people comply with coercive, utilitarian, and persuasive power. These types of power roughly relate to prohibiting, financial and cultural instruments respectively. More recently, Boulding (1990), surveying the literature on the subject of power, arrived at a similar distinction. Power may be based on the triad threat, exchange and love; highly correlated to destruction, production and integration. He is the proponent of the stick, the carrot and the hug, symbolizing these three main means of control over others. Again, these three types of power roughly correlate to prohibiting, financial and cultural instruments. He does not mention the structural instruments separately. The main types of mechanism here thus correspond roughly to the main means of power. This does not mean much more than that the instrument development here is not at variance with main approaches in political theory.

There is a vast body of literature on specific instruments actually or potentially applied. However, there does not seem to exist a comprehensive survey of the instrument field that might help the design of instruments for environmental policy.

2. Administrative theory
From the often very practice oriented administrative literature a few elements have been chosen. First, some observations are made on the policy cycle approach, then some remarks are made on the structuralist approach to administrative science. Finally, the situationist approach is treated more extensively, primarily on the basis of different publications of Bressers.

Policy cycle
Administrative theorists generally use some model of the policy cycle, which distinguishes phases and steps. Dror (1971), an influential theorist on policy analysis, has drawn up a list of four stadia: meta-policy making, policy making, post-policy making, and feedback. The four stadia are further subdivided into eighteen analytical steps. Post-policy making, the subject of this study, has three steps only:

◇ decision to implement, and mobilisation of support
◇ implementation
◇ evaluation of actual policy

There is not much analytical insight in policy instruments to be found here. The whole instrument discussion has to fit into "implementation", only one of the eighteen steps distinguished.

Other analysts have come up with large numbers of instruments. One of the longest list of policy instruments ("mechanisms") is by the well-known policy analyst Majchrzak (1984, pp.26-27), after Coates (1978). The main distinction, following familiar lines, is into informational, financial, and regulatory and control mechanisms. Operation, the fourth element, is very similar to government provision, Geelhoed's type 2, that is subsumed here under prohibiting instruments. The mechanisms themselves are practical and quite devoid of any theoretical underpinning, see the list in table 2.4.3. They indicate the complexity that may be put into instruments, based on real and important elements. In policy design such complexity would make any rational choice impossible. Main distinctions are to function in main decisions, with hierarchically subordinate decisions made later, aided by specialists in the fields concerned. That is one reason why this list

TABLE 2.4.3 POLICY MECHANISM AS DISTINGUISHED BY MAJCHRZAK (1984), PARTLY BASED ON COATES (1978), SLIGHTLY ABRIDGED

1. Information related	**2. Financial measures**
▸ Generation of information by means of: #data collection #technology assessment #demonstration #public hearings #evaluations #monitoring #research and development ▸ The packaging of information #as curriculum #display of pros and cons ▸ The dissemination of information in terms of: #reports #extension programs #seminars #conferences, symposia #trade fairs #state technical services ▸ Stimulation of interest #education #providing a forum #publicity #propaganda #fear and threats ▸ Withhold information ▸ Proposing model legislation	▸ Taxes: #value added tax #excise or income tax #corporate or personal tax #tax write offs or subsidies #depreciation and depletion allowances ▸ Grants ▸ Contracts ▸ Loans ▸ Rewards for innovation and invention ▸ Incentives (e.g., matching funds, scholarships, grants) ▸ Earmarking funds, setting floors and ceilings ▸ Insurance of loans, crops, investments, etc. ▸ Compensation for loss ▸ Underwriting ▸ Priorities on funding ▸ Allocation of funds
3. Regulatory and control measures	**4. Operation**
▸ Regulate/deregulate ▸ Ban ▸ Legislate ▸ Require warranties ▸ Set standards ▸ Zone ▸ Certify ▸ Monopoly privileges ▸ Licence ▸ Form interstate compacts ▸ Codes ▸ Cease and desist orders ▸ Grant rights ▸ Inspection requirements ▸ Institutionalize ▸ Audit ▸ Rationing ▸ Quotas ▸ Limit liability ▸ Prohibitions ▸ Import ▸ Export ▸ Copyrights ▸ Patents ▸ Eminent domain ▸ Declare martial law ▸ Government control or monopoly ▸ Registration and mandatory reporting ▸ Fines and punitive damages ▸ Court decision, injunctions ▸ Substitute criminal for civil sanctions or vice versa	▸ Building civil works ▸ Building facilities (e.g., drug treatment facilities) ▸ Operating facilities (e.g., traffic control systems) ▸ Establishment or support of an industrial base by government purchase ▸ Demonstrating **5. Policy related function** ▸ Setting of policy ▸ Defining priorities ▸ Setting objectives ▸ Delaying decisions ▸ Coordinating affairs

can have only a limited function, e.g., as a checklist for ideas. A second reason is that the list is not a typology or classification system since its categories overlap. One may use incentives, e.g. grants, its subtype, while grants are a separate category of financial measures themselves, as is the allocation of funds, while at least partial overlap with subsidies might also occur.

Mitnick (1980), more concisely, distinguishes two main types, with eight subtypes to cover the field, see table 2.3.6. His main distinction in directives and incentives coincides with prohibiting and financial instruments, in principle. In filling in these main types, however, he uses several dimensions. The first directive type instrument is public enterprise. Public enterprise (Geelhoed's type 2 or 7), has as dimension the nature of the owner of facilities, not a working mechanism. It is therefore not an instrument type distinguished here. Common law versus administrative rules or standards is a distinction based on a main categorization of the juridical status of instruments which exist in Anglo-Saxon countries only. As explained before, this does not make much sense in a non-Anglo-Saxon world nor in terms of the working mechanism of instruments. The first two types of incentives seem to be distinguished by the target of the instrument, some economic object for tax incentives and an environmental target for effluent/user charges. This dimension is not applied to subsidies; these mirror the direction of payments as compared with charges only. A promotional campaign would be a cultural instrument whose nature is yet clear. Laissez faire, the fifth and last type of incentive seems to exclude any regulatory means and thus cannot be a type of instrument.

Structuralists

Two German social scientists have been most influential in the Western World in developing the systems approach for administrative sciences, Luhmann (e.g., 1968) and Mayntz (1984). Their main appeal[1] is their specification of functional requirements on governments and government bodies that may be incorporated in "neutral" empirical analysis. Both stress the symbolic character of social communication and the procedures required to maintain institutions. However, they add no instrumental analytics for specific regulatory areas, i.e. the environment, involved.

TABLE 2.4.4 REGULATORY MEANS AS DISTINGUISHED BY MITNICK (1980)

Regulation by directive	Public enterprise (extreme case)
	Common law
	Administrative rules or standards
Regulation by incentive	Tax incentives
	Effluent/user charges
	Subsidies
	Promotion campaign
	Laissez faire (extreme case)

Situationism

Many administrative theorists stress the fact that there are no "best" instruments, that one has to look at every occasion of policy formation separately. Knowing the situation, one can then develop the best instruments, and especially the best instrument mixes for that specific policy problem. Such an unstructured approach would surely lead to chaos. Bressers, taking this position, developed a model that might clarify which aspects of the dynamic situation can be relevant, see his PhD dissertation and especially Bressers and Klok (1988). His process approach covers the full functioning of all related public and private societal processes. In my terminology he covers the governmental model leading from decision to implementation, the interface government-society, as well as the societal model indicating the direct effects of policies. The approach is actor oriented. Processes are an ongoing string of decisions and interactions, each based on subjective, interactively

[1] Their style of writing, especially that of Luhmann, is not appealing. Maintz is a translator of Luhmann.

assessed pictures of the situation and of the intentions and powers of all related actors. Each process may be described by an internal setting in terms of about thirty items that usually will not all be relevant for some of the processes studied. Added to these, are the external circumstances in which the process functions. The situation is simplified by assuming a one-actor choice situation, both at implementation side and the side of the regulated process in society. Instrument choice itself may be one level in what has thus become a self-referential analysis.

The main types of instruments are related to the five different ways the decisions of an actor may be influenced, with some examples given in brackets:
◇ Increasing the number of alternatives available (technology development)
◇ Decreasing the number of alternatives available (through physical measures, like blocking a street)
◇ Changing the advantages and disadvantages of alternatives to the actor (it is here the instrumentality of most current forms of regulation lies, ranging from general bans to conditional bans and permits, and including economic instruments such as price regulations, levies, taxes, subsidies and tax facilities)
◇ Changing the normative weights attached to the different outcomes of alternatives (normative persuasion)
◇ Adding information on alternatives and their effects (general information, individual counselling)

When trying to fit these five entries to changing an actor's behaviour into the four types of mechanism framework, some striking similarities and differences arise. Normative and factual information, two main types of cultural instruments, are identical to the latter two in the above list, changing normative weights and adding information. The first two, increasing and decreasing the number of alternatives, do not fit into the approach chosen. They are subsumed under either of the regulating mechanisms, in the same way as Geelhoed's "state procured facilities" was. In situationist terms, these activities have actors that are to be influenced by "changing the advantages and disadvantages of the alternatives to the actor". Thus, all instruments apart from the cultural type, the prohibiting, financial and structural instruments, fall within that latter category of changing advantages and disadvantages. The prohibiting and financial instruments are present in the examples given by Bressers. Structural instruments are mentioned as well, e.g. liability. There is no inherent system or guiding factor in categorizing instruments within his single category of changing advantages and disadvantages.

The search here for elements for more macro instruments is not successful. In the one-actor analysis, the differences in level of aggregation of application within each main type, the instrument's macro character, cannot even be stated. However, Bressers generalizes and distinguishes, e.g., between information of a general nature and individual counselling. Here, the level of *macro-ness* is at least implied. The main function of introducing his analysis, however, is confrontational. What are the differences between the situationist approach to policy instruments and that developed in this study?

First, in this study instrument choice is not itself modelled. More important, modelling administration and modelling society is seen here as an activity subordinate to a quite abstract, general instrument analysis, not parallel to it and certainly not integrated into it, as situationists seem to prefer. Only after main decisions have been made in selecting instruments in terms of the two interfaces, based on very general knowledge, would it make sense to investigate government and model society and work out the details of implementation. The instrument to be specified or the instruments to be compared, must be determined first in terms of the interfaces involved. Details, based perhaps on dedicated models, are filled in later, in the preparation of implementation. In the situationist approach, however, instrument choice in environmental policy is not a selection procedure to start with, but the final result of broadening mixtures of policy instruments.

> *"Both governments and the target groups of policies will have to learn by effecting what forms an effective approach. It is also important for that reason to apply several change strategies simultaneously .... Thus, in environmental policy even more than elsewhere, it is senseless to search for 'the philosopher's stone', the single, ideal instrument for policy." (Bressers 1991, pp.16-7, translation G.H.).*

The approach adopted here, by contrast, is that a very limited number of instruments, working as broadly as possible, should do the job. I am thus looking for only a limited number of philosopher's stones.

Still another difference is that the situationists tend to define instruments in terms of the most important mechanism. In these terms, technology development is an instrument for applying environmentally better techniques in industry. Here, such a concrete activity could not be considered an instrument but rather a social process that might be influenced by an instrument, such as general taxes on primary emissions. Here instruments are defined in terms of their primal working mechanism only, an approach which allows different choices in the level of complexity in models of society, independent of instrument definition. What the influence on technologies used is, will thus be primarily an empirical problem. Data, let alone accepted theories indicating the causes of technology development, are extremely scarce. A tax, taxes only what it taxes, according to the interface chosen. The fact that substantial taxes are certain to influence the direction of technology change is an argument to introduce them. A specific technology development itself hardly can be an instrument of environmental policy in the approach developed here. It certainly would not be a macro instrument.

Finally, when starting at the level of the individual actor model, the modelling required for macro instruments is very complex, too complex ever to allow their systematic development. Simple economic theory, for example, is based ultimately on the same subjective rational actor model. However, these basics are remote when the elasticity of supply and demand functions in a micro-economic model, let alone a macro-economic model. If different types of actors in broad sectors of society are affected, the actor-oriented process model is too complicated to form the basis for even case oriented policy design[1].

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[1] However, Bressers generalizes from his findings quite broadly, easily surpassing the practical limits of the individual actor model. He is a main proponent of financial instruments in the Netherlands. One of his famous statements is that firms are not against financial instruments because they are less efficient than command and control regulation, but because they are more effective since as they cannot be evaded.

3. Law and welfare theory

There is a vast literature on *law as an instrument of social change*. Especially in the United States "instrumentalism" is a dominant approach in progressive sociology of law; see Schuyt (1985) for a critical analysis of pros and cons of instrumentalism. This instrumentalism is opposed to the formal approach of law as a hierarchical, partially normative, ordering system. The instrumentalist analysis is not very helpful in distinguishing instruments. The position taken here, is that in modern societies, policies usually must be embedded in some formal decision-making structure, which defines their procedural juridical status, and they have some formal juridical status as to the type of rule applied[1]. The system of law defines boundaries and provides opportunities. With appropriate changes in law, all the instruments discussed so far may be given some judicial status which makes their functioning possible within the formal juridical framework of a specific society. For the working mechanism of prohibitions in permits, for example, it need not make much difference if fines on non-compliance are administered through administrative law or through penal law. For the actual practice of enforcement this difference is significant, of course, since it decides which officials are involved.

Some authors use the *formal juridical status* of policy instruments as a primary basis for the categorization of policy instruments, see Mitnick above. As the formal status of the prime mechanism is often relatively unimportant, this status does not give much guidance in policy design. A formal status would not collectively cover instruments whose primarily relevant aspects make them similar, such as fines in administrative and penal law. Conversely, instruments that are intuitively seen as very similar, may belong to very different juridical categories. Thus, WRR (1992) has to use the newly created juridical category of "transaction instruments" to collectively cover emission taxes and tradable emission permits.

Lawfulness as a metaphysical entity, indicating i.a. legality and justice, does not play an independent role here. That role goes to the political decision procedure on the one hand, outside the domain of this study, and on the other hand is partly reflected in the principles for instrument and policy design and evaluation, in Part Three.

These contributions from the sociology of law and the formal status of public activities do not seem very helpful in developing a framework for instrument classification. A more important contribution of law lies at the boundary of *the philosophy of law* and *welfare theory*. The principal authors followed here are Calabresi and Melamed (1972) for the law side, Baumol and Oates (1988) for the adjoining welfare theory, and Coase (1960) who takes a boundary position. This quite fundamental analysis relates to the structure of society, at the level of "rules that shape rules", as a hierarchical system of classification.

Central to this structural analysis is the assumed nature of environmental problems, with two main entries from welfare theory applied to the analysis. First, environmental problems can be seen as an overconsumption of collective goods. Collective goods are

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[1] This is the case even if no formal status in law is specified. As soon as an incidental subsidy, for example, is applied regularly, rules on its application develop, if only derived from practice, on which rights of others may be based. Regularity in public behaviour would thus already bring this behaviour into the realm of law.

defined in terms of two characteristics[1]. The first is that the increased consumption of one unit by someone does not diminish the availability of the good to others, i.e. *undepletability* or *jointness of supply*. The other element is that if the goods are supplied to one, they are available to all, i.e. *non-excludability* or *non-tradability*. Environmental problems can also be viewed as technological[2] external effects, that is the physical effects of one's activities on others that are somehow not reflected in market prices. Both views are true, in the sense that environmental externalities are usually effects on collective goods. Thus, emissions to the environment may alternatively be seen as the use of the collectively owned environment ("you use our clean air for your hazardous emissions"), or as external effects, producing a negative collective good, i.e. a collective evil as a negative collective good ("the hazardous emissions of your factories are detrimental to the health of us all"). Solutions may result from both lines of enquiry.

How those solutions are shaped is related to the primary distinctions of Calabresi and Melamed (1972) on how entitlements can be created, see Barzel (1989) for a survey of the discussions on this approach that followed. They distinguish three types of rules that each maybe specified in two symmetrical variants. The first variant protects entitlements on not being disturbed by activities of others, the second, symmetrically, protects entitlements to disturbing activities. Six alternatives emerge, all formulated from the point of view of the potential polluter (or otherwise environmentally harmful subject), see the table.

TABLE 2.4.5 SIX MAIN TYPES OF ENTITLEMENTS RELATED TO ENVIRONMENTALLY HARMFUL ACTS

Property rule 1:	I may not pollute unless those polluted allow me to.
Property rule 2:	I may pollute and will be stopped only if compensated for doing so.
Liability rule 1:	I may be stopped from polluting but must then be compensated.
Liability rule 2:	I may pollute but must compensate those harmed.
Inalienability rule 1:	I may not pollute and cannot acquire that right.
Inalienability rule 2:	I may pollute and cannot trade that right.

These six options are mutually exclusive, in principle, at a case level where they ultimately apply. For the domain of environmental problems a firm choice for one approach would be most clear. If that is not possible, subdomains would have to be delimited, with one option applicable in each subdomain. Since activities are not exclusively environmental, there will also be boundaries with entitlements for non-

[1] Several slightly differing definitions exist, see Mishan 1968, Sen 1969, Hennipman 1969, and Baumol and Oates 1988. Government supplied goods are also called collective goods. These may, or may not, be collective goods in the welfare-technical sense of the term used here.

[2] As contrasted with pecuniary ones, see Baumol and Oates pp.29-32.

environmental aspects as well. Limiting the number of boundaries is the be most practical course.

Inalienability, the most extreme property right, does not appear acceptable as a general approach, in either version. Either virtually all economic activities would come to a halt, or no limits on the harm inflicted on others could be enacted or negotiated. It might be applicable for specific situations, however. A right to not to be polluted, rule one, could make sense for very dangerous substances that can be avoided. An inalienable right to pollute, rule two, could make sense if stopping pollution would necessarily lead to unacceptably high costs. Extreme examples are the full prohibition on the emission of genetically modified human pathogens, as inalienability rule one, and the right to human bodily emissions (CO_2; CH_4; etc.), as inalienability rule two.

The four remaining cases are formed by a pair granting the right to polluting and a pair granting the right not to be polluted. In 1960 Coase proved, under some quite stringent assumptions, that for reasons of efficiency a choice between these two pairs (he did not distinguish property rules and liability rules) cannot be made. His main assumptions are that there are no transaction costs and that there is full knowledge of all environmental effects by all parties concerned. The optimal amount of damage prevention then would be brought about regardless of the prevailing type of entitlements. His famous article led to the treatment of external effects on collective goods abstracting from specific property and liability rules, or at least assuming an open choice to be made in terms of whose rights are to be protected. When the basic assumptions are not fulfilled, the choice is no longer neutral as to the amount of damage prevented, neither choice being optimal. The choice then becomes relevant to how efficiently the damage prevention is realized. Calabresi and Melamed (p.1118) describe the conditions under which either type of rule is most efficient. Generally, the most efficient choice gives the property right to those whose costs of damage reduction is lowest. If the transaction costs are high, and if it is not clear in advance whose damage prevention costs are lowest - a situation quite usual in environmental affairs - they take the liability approach:

> *"We are only likely to use liability rules where we are uncertain because, if we are certain, the costs of liability rules - essentially the costs of collectively valuing the damages to all concerned plus the costs in coercion to those who would not sell at the collectively determined figure - are unnecessary".* (p.1119).

They see the choice of where the entitlement is to lie as still open, especially if a choice is not only to be made on efficiency grounds but also on other grounds, such as those involving distributional grounds.

Quiggin (1988) distinguishes private property rights and common, or public, property rights as well as the situation of *open access*. In this latter case he wrongly maintains that no property rights are established at all. Bromley (1991) fulminates against all those that blur this by now well-described distinction between open access as non-property (res nullius) and public property (res cummunes). Hardin's "tragedy of the commons" is especially responsible for propagating the confusion between public property rights and open access. The failure to distinguish between public property and no property inevitably points to private property as an, unjustified, sole solution to the problems occurring. Bromley, in his (1978) bases his position on Calabresi and Melamed. Open access may result from three situations, in terms of Calabresi and Melamed. It may result from property rule one, but with the collectivity not guarding its property rights, it may result from liability rule one where the open access deliberately is given to those using the environment. Hardin's solution, to establish private property rights, thus is one option only out of several. The other options are to better effectuate the collective ownership under property rule one, or to protect the environment through liability rule two. These two fundamentally differing options, both not taken into account by Hardin, are reflected in the tradable emission permit and the emission tax respectively.

Many authors restrict their analysis to liability protecting the pollutee, that is liability rule one, the soft protection of the pollutee. Such authors are now confronted with technical arguments indicating that a choice either way is not yet due and should depend on better knowledge of the actual functioning of society. Bohm and Russell, for example, point out that (p.434):

> *".... writers on liability seem, rather surprisingly, to have ignored the problem of damage-**seeking** (emphasis G.H.) behaviour..."*,

that would occur if liability for environmental damages were established. One middle position taken by many authors is that if damages are paid by the parties causing them, as one way of internalizing the formerly external effect, that these payments at least should not go to the victims. That mixed position, no meat and no fish in terms of Melamed and Calabresi, seems not well-founded,see below.

Liability in private or common law, however, has already been built on a choice in this respect, that people should remain undisturbed by activities of others as much as possible and that if damages occur these should be paid to the victims by those causing the damage. That I may not throw my refuse in your garden as I please is not just a matter of social efficiency. It always has been or has become one of social justice as well. The secondary inefficiencies of this choice, like damage-seeking behaviour, have been known for a long time. They have been remedied in private law to some extent[1]. Still, under current tort law, abuse is perfectly possible. People may get their car reconditioned by having it damaged while seemingly being in the right. Nobody would consider this a reason to pay all damages to cars to the government instead of to the injured party. Such fraudulently acquired, unjustified advantages may be prevented by the potentially injured party through driving carefully, by the threats of criminal law, and especially by refinements in all rules pertaining to property and liability. The two thousand year old Roman principle of *bona fides* would preclude such advantages, even if they are not fraudulous.

Baumol and Oates and Calabresi and Melamed all consider the implementation of liability through private law and through some collective provision. The former two restrict their further analysis even to the collective case, where liability payment is effectuated collectively, as through taxes, and is not paid to the victims, in order to prevent damage-

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[1] In the most often cited example of noise compensation near airports those seeking the noise in order to receive benefits are not compensated but overcompensated. Not compensating, to create an incentive towards damage prevention, is advocated for example by Mishan and Baumol and Oates. However, if the airport wants to prevent damage payments it can do so by buying the property where people would otherwise build their homes and receive compensation.

seeking behaviour and to promote damage-preventing behaviour. That option, which requires specific implementation by government, will be treated here further under the main heading of financial instruments.

The question remaining is what the role could be of non-collective private liability, and perhaps private property rights as well. Clearly, the role of private liability is restricted to damages that will not emerge more than several decades after being caused and damages that somehow may be connected to the parties causing them. These latter restrictions on the applicability of private liability are not fixed. In the course of time they change and they can also be changed deliberately, to create a structural instrument. The role of property rights seems primarily limited to semi-collective goods where one may be excluded from consumption (also seen as a subtype of collective goods), natural resources being the main example. The possibilities for private property rights to such semi-collective goods will not be worked out further in this study. Examples are hunting and fishing rights, tree harvesting rights, and exploitation rights to mineral resources kept or sold by government.

As far as long-term perspectives on liability developments are concerned, Pound (1921, pp.185-9) is still illustrative. He describes, and defends, the ongoing transformation of law from extreme 19th century individualism towards the incorporation of collective exigencies[1]. The individual (or collectivity) injured through environmental mechanisms caused by others is and should increasingly be entitled to liability payments. The central question is how a rich and healthy environment may be practically protected by liability rules. One extreme, going in the direction of property rights, has been enacted in the Michigan Environmental Protection Act see WRR (1992), based on Bocken (1969). Any private person may sue any company for any "unnecessary environmental damage". The nature and extent of the damage and whether it is necessary must be decided in court, with judges functioning as ad hoc administrators. This very general state law seems to have been superseded by later, more specific, environmental laws and regulations, especially federal ones. As an example of a structural instrument, extended liability will be worked out below.

In many discussions on liability, as in those on compulsory liability insurance, the problem of financing the collective cost of damages is central. The analysis here is fully directed at the preventive effects of liability, in a line of reasoning also taken by Bardach and Kagan (1982b). In the tradition of quantified economic modelling of simplified situations liability has hardly played a role and is thus not usually included as an alternative in the analysis of policy instruments. A recent exception is Kitabatake (1988).

4. Economics

Economists have achieved the most in preparing the replacement of the still dominant command and control types of instrument with other, primarily financial, instruments,

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[1] Pound 1921 states for example: ".. there is a very marked tendency in judicial decision to regard the social interest in the use and the conservation of natural resources; to hold, for example, that running water and wild game are, as it were, assets of society which are not capable of private appropriation or ownership except under regulations that protect the social interest." (p.189).

designed specifically for environmental problems[1]. The literature ranges from general surveys of instruments, to the comparison of two, or three instruments in specific, empirically or theoretically modelled situations. Here, the comparative treatment of instruments by Bohm and Russell (1985) will be surveyed first. Their instrument analysis is micro-economic in character, even if placed in the setting of full or partial equilibrium analysis. Their analysis is restricted to what they see as potentially useful instruments. Thus they do not explore the possibilities of general subsidy instruments, for example. Next, the possible role of macro-modelling in instrument analysis will be reviewed briefly.

Within these boundaries, Bohm and Russel (1985) survey the entire discussion on instruments taking place in economics. They distinguish the following main types of instruments (reversing their sequence):

1* moral (per)suasion
2* direct intervention related to the specification of behaviour
3* deposit-refund systems, performance bonds and liability rules
4* administratively defined tradable commodities or rights, with price forming left to the market
5* administratively set taxes or prices

This classification might easily be transformed into the four main types distinguished here. It seems wiser however to add, provisionally, some of their major distinctions. The first two headings pose no problems. Moral persuasion is one element in cultural instruments and direct interventions are the main type of prohibitions. I subdivide the third somewhat inhomogeneous group. Deposit-refund systems become a subcategory of financial instruments. Performance bonds are not an independent instrument but may be added to other instruments. Liability rules are a main item under structural instruments. Tradable commodities and rights, in 4*, lie at the boundary between prohibitions and financial instruments, e.g. it is forbidden to emit, a prohibition, if one has not first acquired a permit, i.e. a financial instrument. As the working mechanism is primarily the prohibition, I treat them under prohibiting instruments. Admittedly, this choice is somewhat arbitrary. Finally, administratively set taxes and prices are now the second subcategory of financial instruments. Instruments as treated by Bohm and Russell thus will be handled here under the following six headings:

◇ structural instruments
◇ cultural instruments
◇ financial instruments 1: taxes and pricing
◇ financial instruments 2: deposit-refund systems
◇ prohibiting instruments 1: tradable permits
◇ prohibiting instruments 2: direct interventions

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[1] There is a general feeling among economists in several professions, that environmental policy should move towards financial instruments, see Frey et al. 1984 for an international survey of 1400 economists. One group forms an exception, i.e. the left wing, among publicly employed ecomists in countries with a strong direct regulation tradition. How this might be carried out politically is deliberately omitted here. See Kelman 1981b, however, on the further succesful introduction of these instruments, based on empirical research in this field. It is vital, in his view, to first convince environmentalists of the necessity of these instruments.

Several instruments of the same type will be classified if possible according to their level of aggregation, i.e. to their macro character. Sometimes instruments are not fully defined as to the interfaces they are built from. These then will be extended to a full specification, or several specifications. The resulting list, see table 2.4.6, will thus contain a few more instruments than treated by Bohm and Russell.

Structural instruments

Strict liability, e.g. a combination of liability without negligence and chained liability, is the principal instrument. Bohm and Russell see its main application to activities with unknown chances on scarcely predictable long-term effects, that could hardly be regulated *ex ante* with other instruments (p.434). It seems, however, that real long-term effects, such as those of CFC-generated global warming which extends for centuries, can hardly be covered by liability in any serious way. Liability would apply to the financially definable risks of one process, or group of processes. These cover only a small range of all the potential effects of all types of interference. The society-environment interface, in this case, would be extended beyond problem modules, to full effects and their monetary valuation. Such effects would usually be based on local circumstances and be assessed site-specifically. This instrument therefore lies outside the main domain of this study. The chance that strict liability will lead to effective payment, in case of damage, could be improved through the combination with a performance bond or insurance policy (p.434). This example belongs to a second type of liability.

This second type of liability is the mainly company- or at best, sector-specific liability. There, for example, damages may be assessed according to predefined rules for quantifying the damages, as stated in a contract. Starting operations under such rules implies the acceptance of liability for the damages thus defined. There are examples of this micro approach to liability in the US, e.g. in offshore oil exploration and exploitation (p.436). Combined with a transfer of the liability to a bank or insurance company Bohm and Russell name this instrument "performance bond". They treat the performance bond as a variant of a deposit-refund system, quite different however from the usual ones on products or the ones on hazardous substances advocated here. In a performance bond the transfer of liability has to be paid for in advance, based on the combination of the risk and the amount to be paid, and might be repaid if no damage payments occur. See for further comments on this instrument below under "financial instruments 2" where deposit systems are treated. A more aggregate type of environmental liability will be introduced in the next section.

Cultural instruments

Moral persuasion, including the active provision of information, is seen by them as a case-specific instrument: to prevent forest fires, to buy phosphate-free washing powders, or to return mercury and cadmium batteries separately (pp.453-4). Moral suasion then might be combined with creating facilities, as in the separate collection of batteries. According to the authors its place is mainly, but not exclusively, where other instruments are not applicable. Their examples partly relate to disturbances, like causing fires. Such disturbances will primarily be disregarded here, however. The substance related examples all concern a one substance-one product interface, in a partial life cycle context. The gist of their treatment is that cultural instruments may be worthwhile where other instruments do not apply at all and that their role is a supportive one only, when other instruments are available.

They correctly point out that moral suasion as an instrument should be distinguished sharply from the situation where moral arguments play a role in decisions by producers and consumers already. In the next section, very substantial increases in the level of aggregation of cultural instruments will appear feasible.

Financial instruments 1: taxes and pricing

Bohm and Russell specify two main types of *taxing instruments*. One is a regionally uniform emissions tax, set at a level so as to reach a uniform ambient standard, such as an emission quality. The latter aspect is only a consideration in setting the level of the tax, not an element of the instrument itself.

Number two is an emission tax, or exposure charge, that for each source adds up to a specific tax level per unit of the substance emitted, different for each source, but uniform for each unit emitted. How might this instrument be defined in terms of objects and targets? At first glance it would seem that here the emission would be the target, since the total emission effect of several sources is taken into account in developing the tax levels. On second thought the question arises as to which object in society this tax is bound. The translation of emissions at several places into the source-specific level of the tax takes place administratively, as an internal governmental process. The agency goes to each firm, tells the firm the amount to pay per unit emitted, measures its emissions, and collects the tax[1].

In terms of targets there is thus no difference between the two taxes, they both concern "emissions of a substance". The types of object are also the same: emitting processes. The uniform tax is more macro however, since its objects are "all processes that emit the substance", together, while the differentiated tax has as its objects the emitter that contribute to the emissions at different locations. In both cases the addressees, the owners of the firms as objects, have to be approached separately.

Subsidies have not been treated by Bohm and Russell, as they cannot be relevant in environmental policy[2].

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[1] This instrument is the most statically efficient one modelled by economists. Interestingly, its advantages as compared to the uniform tax are due nearly exclusively to the fact that the tax leaves some areas less polluted than the standard for maximum pollution. The uniform tax reduces emissions "too much"! Dales 1968 already refuted this optimal tax, and not only because it cannot be implemented practically. He took great care in explaining that equal pollution everywhere is highly undesirable (pp. 88-92). The same distinction between emission and immission is discussed quite extensively in the context of tradable emission or immission permits. Atkinson and Tietenberg 1982, p103, when comparing the tradable emission permits with tradable ambient permits, state as a disadvantage of the emission permits that *"the level of allowable emissions has to be reduced sufficiently to ensure that the air quality at the most polluted receptor location meets the standard. This results in substantial over-control of the more distant sources, thus increasing the costs of compliance to achieve specified air quality levels"*. Nichols 1984, pp160-1, favours the exposure charge quite unconditionally, seeing practical problems as the main disadvantage. Russell 1986 in a more neutral way, states that uniform taxes are second best compared to emission charges or emission standards tailored to each source, also without discussing the advantages of realising better quality in many places, while exceeding minimal standards nowhere.

[2] See Vermeulen 1992 on the unsuccessful implementation of environmental subsidies in the Netherlands.

Financial instruments 2: deposit-refund systems

Bohm and Russell describe two variants of government initiated obligatory *deposit-refund systems*, i.e. the use of such systems as an instrument. One is placed on products, because of assumed hazards if these products are not returned separately (p.431). There is no specific relation between deposit level and environmental interferences. Such an obligatory deposit system is not an instrument of environmental policy in the strict sense, since the targets are the products themselves. Addressees are the firms that market the product in question. Of course, such deposit systems may also function without governmental intervention, e.g., if the value of the returned product is great enough. An example is returnable glass milk bottles. This latter deposit system is not an instrument of environmental policy at all.

The other variant places a deposit on chemicals, to be repaid if these are returned to a specified type of processing firm (p.432). The target is related to the chemical substance twice, once for deposit payment, when the substance becomes a potentially emitted one, and once upon refund, when delivery at a processing plant indicates that the former owner will not be emitting that amount of the substance. The ultimate addressee are the owners of a potential emission process, the object of the instrument, and the mechanism is financial. In another publication Bohm (1981, see also his 1981b) comes nearer to the substance-deposit system as will be developed here. Bohm (especially pp.141-9 in the example of application of CFCs in refrigerators, in chapter 6) describes characteristics of the chemicals deposit that allow variants not deemed acceptable here. These are the following:

◇ levels of deposit and refund need not necessarily be the same (no clear relation to emission tax)
◇ deposit-refund systems specify each application of the substance separately (less macro than possible)
◇ differentiation in levels of refunds should be based on characteristics of returning and reprocessing costs for each subgroup of equipment (system boundaries not broad enough).

Another still more specific difference is that reprocessing for reuse is seen as a basis for refunding (p.136). As will be treated extensively in the case of cadmium in Part Five, reuse should not be the basis for the refunding of deposits.

Four years later, Bohm and Russell make similar remarks on the deposit-refund system for chemicals. One is that no equality in present value is necessary between deposit payment and refunding, an approach they propose (p.429). The theoretical implication is that the link between emission damage and (implicit) payment is loosened. The important practical implication is that it is then not necessary to know when the deposit of a returned amount has originally been paid. A second main distinction is related to the objects of the deposit system proposed. It is not one level of deposit and refund for each producer, or group of producers, or group of users, or type of product, as contemplated by Bohm and Russell in various combinations. It is any process that brings the substance into the economy of an administrative unit as a whole, for deposit payment, and any process that removes it from the economy without concomitant emission, for deposit refunding[1]. Repayment thus does not take place at delivery to a processor, but when the substance is processed out of the economy. A final related difference is that refunding also takes place when the substance, in any form, is exported. All differences with Bohm and Bohm and Russell relate to specifications at the society-environment boundary. They take a position at that boundary but then only give some variants and examples that are possible technically. They do not treat that interface systematically and thus do not come up with the macro option of "flows of a substance", as the target, through "all processes in the economy", as the object.

On the further development of deposit-refund systems Bohm and Russell state:

> *"Furthermore, we believe it worthwhile expanding the fields of application contemplated for such relatively unexplored instruments as deposit-refund systems." (p.455).*

Notwithstanding the differences stated, the substance deposit as developed here is doing just that, in their spirit and basically along their lines.

Performance bonds may be seen as a sort of deposit-refund system, as they do (p.432). The typical situation they have in mind is that of a process with potentially hazardous interferences of a not-yet fully defined nature and amount. Payment is based on the potential damages caused, with a return if the activity stops and no damages occur. This can be seen as a damage deposit. If there is a long lapse of time between the start of a potentially hazardous activity and the end of it, a financing problem is involved. This could be solved by allowing the firm not to pay its deposit, on the condition that the obligation to pay if hazards occur is transferred to a financially trustworthy party. That party then would assess the chances of payment and charge the firm a lump sum or yearly payment, based on the chance and expected amount of potential payment. This then is a fluid continuum towards liability insurance.

I would prefer not to treat this third instrument as a deposit-refund system, since in effect no deposit need be paid at all. I would see it as a firm-specific (or sector-specific) type of liability, with the financial level of damages specified in advance. Through the specification of the effect chain beforehand the liability becomes "extra strict". This was the case in the example mentioned above of the liability for oil spills from offshore exploration and exploitation. Liability may be insured as a private activity not related to the instrument. However, insurance for such specific liability might also be made obligatory, involving the transfer of the liability to such a company. This is how Bohm and Russell seem to define performance bonds as an instrument (p.433). The latter addition generally would not be attractive for reasons of regulatory effectiveness. It would subtract much from the regulatory advantages of liability, or of damage deposits, as stated. The incentive to prevent damages through all sorts of decisions, e.g. in product and process design, would be lost, at least partially. That preventive function would remain if the insuring company would retain a full right of redress on the insured company. Effectively the obligatory insurance would then only safeguard payment to the victims. The obligatory insurance would not subtract from the regulative effects. As the

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[1] One practical problem they see when applying the deposit system to a firm, is that the deposit could be fraudulently refunded for an amount of the substance that never has been the object of the deposit payment (p.432). If any amount of the substance coming into the economy is the object of deposit payment this fraud loophole disappears.

victim often is a government, as with clean-up costs at waste dump sites, such a type of performance bond then would be more of a financing instrument than a regulative one.

Prohibiting instruments 1: tradable permits

Three main variants of tradable permits are introduced. One starts with an excess of permits distributed for a region. It then leaves the system to society where environmentalists may combine to eliminate excess emissions through purchase (p.420). They have disregarded this alternative since enough private funds will never be raised to buy away the excess pollution. Their main reasons for not believing in the effectiveness of this variant are the prisoner's dilemma and the lack of a private social organization to overcome the dilemma.

The second variant also starts with a regionally distributed number of permits, but this time an optimal number (p.421). These permits may be traded between emitters.

In the third variant emission rights are tradable, to the amount the standard states (minus background load). Such emission rights exist for every monitoring point. Emissions at any location can be transformed, by a publicly determined regional distribution model, into contributions to emission at each monitoring point. Any polluter thus knows the amounts of emission rights he has to buy at each monitoring point polluted by him to build up a "full" emission right at his emission location. This second variant can hardly function practically as it seems impossible to establish as many simultaneous markets as there are monitoring points. A major drawback of a uniform emissions standards approach is that a mechanism is set in motion to fill up the "emission space". Differentiated emission standards, e.g. those incorporated into zoning laws, might solve this problem. This would further complicate the functioning of the emission markets.

For substances not transported too far between emission and immission, there is a variant falling between tradable emission and tradable immission rights. It assumes regionally-free tradable emission permits, and either no trade between regions or trade with a transformation factor based on relative cleanness (p.422), e.g. two emission units from a clean area for one in a relatively dirty area.

Tradable rights on commodities, common for food and fuel in war time and in former communist countries, although broadly applicable, have targets too unrelated to environmental interferences to qualify as instruments for environmental policy in the strict sense. Prohibiting such rights in the first place seems quite a heavy measure for an instrument having environmental effects only indirectly.

Prohibiting instruments 2: direct interventions

Bohm and Russel distinguish several variants of "direct interventions", "command and control", or "regulations", all called prohibitions here. They are forced negotiations between polluters and pollutee (p.440), performance standards (p.441), the regulation of decision variables correlated with emissions (p.443), design standards (p.444) and bans on products and processes (p.445).

Forced negotiations fall outside the domain of environmental instruments considered here since their content is purely procedural. No objects and targets are specified. The means of coercion are not specified either. With a certain measure of good will they could be seen as a structural instrument of a low level of aggregation.

Performance standards, or effluent standards, may relate to all substances as emitted by one process, or group of processes. I prefer the term "emission standard". The addressee is one specific owner, as is the case with individual permits, or the group of all those that use the process. An example of the latter is the general limit on the amount of SO_2 emissions per kWh in the German regulations on emissions to air (TA-Luft). In the US these performance standards may differ between old sources and new sources (p.442).

Decision variables are indicators of emissions, especially apt when real monitoring is costly or impossible. Such indicators might take several forms. With cars, it may be tests of new cars in a controlled situation. I will refer to emissions indicated by such "decision variables" as "estimated emissions". These may play an additional role in the instrument strategy to be developed, not as a prohibiting instrument, but combined with a financial working mechanism, such as an "estimated emission tax". If no specific link is made with estimated emissions, decision variable are design standards.

Design standards may specify the technical or behavioral aspects of installations. Bohm and Russell seem to include their application to products as well. Design standards do not specify an environmental target. They specify technical aspects of products and installations and they may specify how these should be used. They are therefore not instruments for environmental policy in the strict sense. Currently, however, they form the main body of instruments for environmental policy[1], in the broad sense.

Bohm and Russell distinguish two types of bans on products and processes, i.e. of general application and those restricted to specific areas. I would prefer not to treat bans as a separate instrument, but to include them in design standards. The reason is that a boundary between them is difficult to define. A general design standard may be formulated so that a certain substance may not be used in it, e.g. cadmium-free stabilizers in PVC window frames as is the case in the US. This, however, is fully equivalent to a ban on PVC window frames with cadmium containing stabilizers. A special case, indicated in the example above, is where the design standard refers to a substance in a product, as potentially hazardous in use or when subsequently reprocessed and discarded. If the product itself is the forbidden substance, as with DDT, it becomes a bit artificial to avoid the word "ban". In an instrument sense as used here, however, ban and design standard are fully equivalent, there as well. General design standards or bans may be area- restricted, as in zoning laws. All instruments having individual installations or types of installations as their object, may be differentiated according to area.

See table 2.4.6 below for a survey of the instruments of Bohm and Russell as interpreted here. The six types are the same as those used in the survey above. Since the deposit system will show a higher potential of aggregation than the taxes, these are placed first,

[1] Even emission-based instruments such as emission standards for installations are in fact often design standards, **aimed** at realising the emission standard. Even tradable permits as exist in the US are generally implemented through design standards that are **assumed** to restrict the emissions to the levels for which permits have been acquired. The variables specified in the standard are then "decision variables".

TABLE 2.4.6 INSTRUMENTS DISTINGUISHED BY BOHM AND RUSSELL (1985), ORDERED ACCORDING TO MAIN TYPES (VERTICALLY) AND LEVEL OF AGGREGATION PER TYPE (HORIZONTALLY)

↓ *Instrument types*	*Aggregation →* *1*	*2*	*3*
Structural instruments	(forced negotiations)*	firm-specific "extra strict" liability,	sectoral (?) strict liability
Cultural instruments	moral suasion, esp. one substance product information		
Financial instruments 1: deposit-refund systems	product deposit-refund system	chemicals deposit-refund system per firm/sector	
Financial instruments 2: taxes and pricing	(emission tax = firm differentiated emission tax on substances from one process/location)*	uniform level emission tax one substance from all processes	
# Prohibiting instruments 1: tradable rights	(tradable emission permits per monitoring site)*	(general / regional tradable emission permits in excess)*	general / regional tradable emission permits (as to reach emission standards everywhere)
Prohibiting instruments 2: direct interventions	design standard for processes and products, including: behavioral rules. Per process type / firm	estimated emission standard: per process / per firm	emission standards for all substances: for one firm / for one type of process of all firms

*Being incapable of implementation, these instruments do not seem practically relevant.

in deviation from the order of treatment above. Per type, instruments have been classified according to increasing level of aggregation.

5. Final remarks on disciplines.

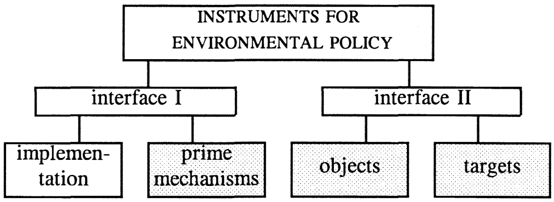
To conclude this longer section on disciplinary thinking on instruments, it should first be pointed out that the contribution of most disciplines does not seem to be substantial in terms of a typology of instruments, neither in terms of working mechanism at interface I nor of object and target at interface II. Only economists have a substantial body of instruments discerned systematically.

A second observation is that, contrary to all other disciplines, virtually the entire body of economic literature is about instruments for environmental policy in the strict sense. Even the permit is usually treated as an emission permit, although that version hardly exists in practice. The targets discussed are, in order of occurrence in the effect chain, emissions, immissions and damages. Environmental problems, such as targets between emissions and damages, are lacking. Other disciplines do not specify, let alone distinguish environmental targets in a systematical manner.

A third observation is that the micro-macro dimension, although common in the field of economic policy, is not referred to explicitly in discussions of instruments for environmental policy. In this respect economists hardly differ from writers in other disciplines.

Finally, the models used by economists to describe instruments and to assess and evaluate their effects are usually of the comparative-static type. They leave the route of change because of the change-over to a new instrument of environmental policy unspecified. Dynamic effects on technologies are not modelled quantitatively.

2.4.4 Main instrument types in their most macro forms



This section effects a first synthesis of instruments, with the working mechanism the prime characteristic, combined with objects and targets aggregated as much as possible. In the next section some observations on implementation will be made. The theoretical analysis of instrument interfaces in the preceding chapter indicated possibilities for much more macro type instruments than those in the survey of Bohm and Russell. The main combinations resulting from that analysis were the problem oriented life cycle analysis of products (LCA) and the substance flow analysis (SFA), as apparently sensible combinations of macro objects and targets. Each is to be supplemented with some working mechanism at the government-society interface to become a full instrument for environmental policy. The instruments surveyed by Bohm and Russell do not relate to these more aggregate LCA and SFA interfaces. Hence the first task here is to investigate which working mechanism, and which further interface specifications, might best be added to those two specific combinations of objects and targets. That choice being made strategically only, the next task is to describe the main possibilities for macro instruments of each type, making an initial list of instruments for macro environmental policy.

Next, the most relevant structural, cultural, financial and prohibiting instruments, in the familiar order, will be chosen and defined. The list of macro instruments is summarized in table 2.4.7 at the end of this section, as the first major result of the design process this study is about.

Working mechanisms for life cycle analysis (LCA)
The life cycle analysis might be combined with structural mechanisms. The kernel of such an instrument would be an institutionalized place for LCA. One could imagine that in certain situations LCA is a procedural requirement for a decision, like an Environmental Impact Statement is in most countries now. Decisions on acquiring products, by (non-environmental) government agencies should be supported by an LCA. Or, claims by firms, in their advertisements, as to the relative environmentally-friendly nature of their products should be substantiated by ex ante LCAs. Similar rules apply to other material aspects of products as well, e.g. the rules governing TV advertisements in many countries. Such working mechanisms seem to require a more developed analytic instrument then is available now. With the maturity of LCA such structural instruments might become more worthwhile.

Cultural mechanisms could easily be connected to LCA. Ecolabelling programmes exist already, with the German "blue angel" as the oldest example. An ecolabel conveys the relatively beneficial environmental effects of the product concerned. LCA could help solve the analytic problems these programmes encounter. From an instrument point of view, a distinction should be made between using LCA in ecolabelling for publicly assessing each individual product, a micro instrument, and supplying a method of LCA-based ecolabelling that may function fully privately at the case level, a macro instrument. LCA-based instruments are reviewed below, under the entry "Cultural instruments: a choice".

Financial mechanisms cannot be connected to LCA in a satisfactory manner, for one reason because of the inherent imprecision in the definition of the object. The precise definition of the function is decisive for the quantitative outcome of the analysis. One litre of milk, its functional use, might be the object of an LCA. "Consumption of one litre of milk" is then one way to define that functional unit, implying the inventory of all processes caused by that consumption. Existing analyses define as the functional unit only the packaging required for transporting milk to the consumer. The difference is the milk itself. Breakage, leakage and decay differ for each packaging alternative, including the milk itself and make a difference in the comparison. Including the milk always makes a difference as to the absolute outcomes of the LCA and thus, a difference in terms of problem taxes. What to include will always remain a matter of taste. Are the glasses used for drinking, the pans and furnaces used for heating the milk all to be incorporated into the analysis for taxing milk based on its LCA? If included, would taxing the furnaces, the pan and the drinking glass, on the basis of their respective LCA results, still make sense? The double taxation resulting can hardly make sense. For comparisons such differences and imprecisions do not matter, as long as they are similar for the products compared. If a tax payment is to be based on LCA outcomes, such differences and related imprecisions seem fatal.

One option could be to use the outcomes of the LCA in a less precise manner, also for taxing. Products in a product group, e.g. defined in terms of a chosen functional unit, may be ranked according to their total problem contribution, especially if some method of evaluating problems could be developed. The ranking might form the basis for taxing. First, the worst products could be forbidden. Then a given percentage of the worst products remaining on the market could receive a tax penalty, for then being worst[1]. If these products disappear from the market the next twenty percent would be taxed, etc. The tax would not vary with the amount and number of environmental problems caused, only with the relative position in the group. Between product groups, there would not be a quantitative relation with the amount of problem caused either. Although a practical possibility, this instrument is no longer considered here since it is hardly an instrument of environmental policy in the strict sense.

Prohibitions based on an absolute problem contribution would have the same problems as taxes, in that the absolute outcomes will remain quite disputable. With prohibitions, however, it is possible to exempt any grey areas from direct interventions. The remaining forbidden *baddies* are then genuinely bad. The most macro application I can imagine is the products standard, related to a specified functional unit. The standard is formulated in terms of the maximum contribution to several problems, or, if combined with a specific

[1] Recently, Mr Krisor, at the European Commission, described this possibility for the use of LCA in a personal communication. His two other categories are the products for which nothing changes and those that receive an ecolabel.

set of weights as values, in terms of its "total environmental problem" contribution. If prohibitions are not based on emissions and consequent problem contributions but on factors supposedly related to that contribution (e.g. a minimum percentage of recycled material in a product) the interface does not have an environmental target at all and the resulting instrument cannot then be an instrument for environmental policy in the strict sense.

Working mechanisms for SFA

Substance flow analysis combines the object of all processes with the target of one substance, or one problem. It specifies the totality of all emissions, as part of the analysis. There are instruments available to cover this totality. Two financial instruments from the Bohm and Russell list could each do the job: emission taxes on all processes concerned and tradable emission permits, with controls on all processes concerned. The substance flow analysis, however, is a much more sophisticated tool whose potentials have not yet been realised. It systematically relates the different flows of the substance through the economy. The substance flow analysis for some substance, at its highest level of abstraction, specifies the different inflows and outflows of the substance in society. Inflow (I = chemical formation plus extraction from the lithosphere) minus outflow (O = chemical destruction plus return to the lithosphere) equals net emissions (nE = gross emissions to the environment minus extractions from the environment)[1], or in a simple formula:

(1) $I - O = nE$

The substance flow analysis, the societal part of it, covers all economic processes in society, but only in one material aspect, that of the substance involved. No symbolic economic elements are concerned, as is the case with life cycle analysis, e.g., in describing product functions ("recreation") and in allocating multiple processes according to their different functions ("air transport of both passengers and freight"), see Part Four.

This fuller SFA cannot generally be split into parts, or groups of processes, where a party responsible for inflow, outflow and emission of an amount of a substance, can then be made the regulatee influenced by some working mechanism. Structural and cultural mechanisms using this richer analysis may thus seem impossible at first glance. Prohibitions could create a partial mechanism at best. A maximum could be set for the inflow, possibly distributed if the rights on inflow are made tradable. At the outflow side no such mechanism seems possible. Substance flow analysis is most useful if combined with financial instruments.

Financial instruments are now restricted mainly to the emission tax. Alternatively, however, the deposit-refund system may be applied to societal inflow and outflow. The deposit could be paid at inflow and the refund at outflow. Essentially, such a substance deposit would create the same effects as a fully applied tax on net emissions of the same

[1] Disregarding accumulations in the economy and imports and exports. See Part Four for a more comprehensive treatment.

amount per unit of the substance[1]. This can easily be seen. If the amount of deposit or tax is t, formula (1) transforms into:

(2) $tI - tO = tnE$

If someone increases his emissions by one unit, an extra amount of emission tax t has to be paid. This increase in emissions requires that either inflow is increased by one unit, also leading to an extra deposit payment for t, or outflow is decreased, with a decrease in the refund payment of t, or a combination of both. A decrease in refund is fully equivalent to an increase in payment. For the emission tax and deposit system to be equivalent, it is necessary that only net emissions should be taxed, alternatively. This means that a payment has to be made for the extraction of a substance from the environment[2], as a case of negative taxation.

The main administrative difference between tax system and deposit system, is that in the tax case the emissions have to be monitored for each emitting process, while in the deposit case the inflow and outflow have to be monitored for the processes involved in these activities. Thus the regulatees of both instrument systems differ, while the objects and target, at the other interface, are completely the same.

The importance of the deposit alternative is that the Baumol tax, as the most efficient tax on reducing total emissions, could become a practical possibility. No substances now exist all the emissions of which can be measured, from all sources. An equal tax on all sources is required to reach the efficiency aim that is the basis of the Baumol tax. For many substances, however, the number of producers is very small compared to the number of emitters. CFCs are an example where this ratio is in the order of hundreds of millions. The heavy metals and phosphates are extracted on a global scale through not more than a few thousand inflow processes and several thousand outflow processes. There are several hundred million separate emission processes. Even if the ratio is much smaller then in these extreme examples, the advantages of monitoring reduced numbers will usually remain. *This application of the substance deposit, not for individual processes, groups of processes or products, but for society as a whole is new in the literature reviewed*[3].

Both the cultural instruments based on life cycle analysis and the financial instruments based on substance flow analysis will be discussed somewhat more specifically here when the possibilities for macro instruments of each main type are treated. There is a fuller analysis, especially of their society-environment interface, in Part Four.

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1 See Mäler 1974, who seems the first to have captured this formal equivalence in principle between the emission tax and the substance deposit system, without specifying the administrative differences involved.

2 In the current discussions on CO_2 taxes the different options for financial instruments are mixed. First, there are proposals to pay for environmental processes in a country, fix carbon contents?, abolish the distinction between environment and economy. Secondly, payment for fixing carbon in economic processes, as in agricultural processes, hardly seems compatible with the indirect taxation of emissions, on fossil carbon in energy carriers, as currently proposed. That carbon tax would constitute one half of a deposit system. Consistency then requires a tax on net emissions of CO_2 i.e.? the separate taxing of all non-fossil carbon emissions. See the case on global warming in Part Five.

3 My first external publications on this subject are Huppes and Udo de Haes 1987 and Huppes and Kagan 1989.

Macro structural instruments: a choice
A first structural instrument, that combines legal theory and welfare theory, is the establishment of ownership of the collective goods constituting the environment through the definition of property rights, to make (at least initially) elements of the environment "government property". To effectuate property rights on collective goods specific administrative measures are required. Prohibiting instruments, especially general prohibitions, might be seen as a type of protection of the collective good, but it is very incomplete. Financial instruments establishing a price for the use of the environment, especially of resources, may be seen as the sale of the collective good. This is especially possible for goods which are not completely collective, where some degree of excludability exists. Such property rules-related financial instruments will not be investigated further.

The other approach is the prevention of external effects on public goods through liability rules. The creation of entitlements, on not to be polluted or, more broadly, on some level of environmental quality then is the policy instrument. Liability is the obligation to pay for infringements of the entitlements of those that want to remain undisturbed by the actions of others. Which possibilities exist? The main development in liability rules[1] is towards easier application. Three inter-related changes make one's position stronger when suing for environmental damages. First, there is now a century of development in which risk instead of fault functions as an appropriate link between the injurer and the injured party. This tendency occurs in most Western countries[2] and could be strengthened to create a liability instrument.
Secondly, chained liability, starting in rules on product liability, is becoming more usual. It no longer is the firm dumping chemicals that is liable, but also the party who delivered to that firm, etc., right back to the producer of the chemical waste. Fault or negligence plays a limited role only in deciding who in the chain has to pay. This element might be added to the structural liability instrument.
Thirdly, there is a tendency towards collective liability. In that case, for example, one polluter may be forced to pay the total damages caused on a site, with a right to redress on the other polluters. In waste dumps contaminated with several chemical wastes that cannot all be attributed to a single specific originator this approach solves one problem in establishing proof and setting the level of the damages. This element might be added as well.

Together these three elements constitute strict, joint and several liability, here further called extended liability. A movement in this mainly environmentally relevant direction of extended liability is the most aggregate structural instrument distinguished here.

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[1] See for a recent Dutch survey of developments Hol and Loth 1991. See Drion (1988) for a pledge of more fundamental reform of liability law and for some differences in the foundations of liability between even very similar countries like France and the Netherlands.

[2] See for example Ganten 1988, who proposses a uniform regulation of these tendencies in Europe by introducing the German 'Law on Liability for Environmental Damage' as the model for Europe. An exception to this trend has occurred in the US. Liability for the contamination of ground water with pesticides used agriculturally is now again limited there to cases of negligence only, see Wetzstein and Centner 1992.

A fourth development, mainly in the US, has much increased the practical effects of strict and several liability, i.e. the tendency to quantify and compensate immaterial damages, as with perpetual headaches or loss of relatives, with very large amounts of money.

One main area where extended liability has increased in the last decades has been the clearing of sites contaminated with chemical waste. One recent Dutch inventory estimated the private type costs of clean-up of known sites at about \$50 billion. On an equal per capita basis this would be about \$850 billion for the USA. In both the US (CERCLA = Superfund) and the Netherlands (Interim Law on the Clean-up of Contaminated Soils[1]) an active clean-up of contaminated sites was started in the Seventies[2] and gained momentum in the Eighties. On a per capita basis, the Dutch programme has been the most successful in terms of costs incurred by those held liable, see von Meijenfeldt (1987). In an evaluation of the American Superfund programme Barnett (1989) sees the programme as an example of implementation failure. He measures the failure in terms of the originally planned number of cleaned sites. This seems a rather limited model for evaluating the effects of an instrument. The broader, indirect effects of charging the costs of clean-up to the firms involved is that the financial risks of dumping chemical wastes have soared. Insurance companies no longer are willing to take these unpredictably high risks, see Muoghalu et al. (1990, p.359). The cost of clean-up not only relate to the costs of the clean-up itself. Litigation costs may account for up to 50% of total costs (Barnett 1989, p.10). Thus, firms are increasingly looking for other solutions. One subjective estimate states that the total amount of chemical waste dumped in the US has dropped in the last decade by 60% (personal communication Barnett).

Extended liability has at least four drawbacks. First, in traditional liability, the proof of fault or negligence was complemented by the normative notion that the specific behaviour deemed liable was ethically wrong. This corresponds to a cultural build up of norms preventing the behaviour, even if efforts and costs are involved. Strict liability can no longer have this effect. The damages caused are simply part of life's risks. It is only a matter of expediency to adopt non-damaging behaviour. Causing damage now has become ethically neutral to some extent already. In terms of the main model variables, the loop from structural to cultural variables is lost. That cultural mechanism was probably a main mechanism in old-fashioned fault related liability.

Secondly, extended liability extends the domain of liability as a costly social activity. In the chemical waste cases handled in CERCLA these costs could be as high as half the total costs for the firms involved, according to Barnett (1989). The effects of extending liability are very complicated and costly as one practitioner, Moskowitz (1989), explains. At a societal level differences in litigation may be very considerable between countries and even within countries. The number of active lawyers per head of the population in

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[1] See Drupsteen 1986 for practical ways in Dutch law for improving the possibilities for receiving payment for damages, aimed primarily at strict liability.

[2] In the US, the clean-up was initially financed by a tax on chemicals. Litigation is seen primarily as a means to supplement the still limited means for the necessary clean-up. Compulsary insurance is one way to ease the burden of financing. The regulative effect is then substantially lost. In these terms, other types of financing may be less costly, see Bocken 1987/88. On the other hand, there is now a discussion to make the chemicals tax that now finances the Superfund more environmentally regulative, see Carlson and Bausell 1987. Both these fundamentally diverging and conflicting approaches seem to lack an overall strategy.

California is ten times as high as that in the Netherlands. It amounts to a substantial part of the total working population, in a highly educated section, that is active in formalizing quarrels that might have been prevented or solved in other ways.

Thirdly, widely applied environmental liability requires the courts and judges to acquire specialised knowledge on economic and environmental processes, and to build up a framework for assessing what is liable and what not. Effectively, courts then become a type of regulatory body with judges as the highest ranking officials, see Bardach and Kagan 1982b on this theme. Courts should not have regulatory tasks and bodies other than courts are more suitable for executing administrative tasks.

Fourthly, even extended liability cannot cover the typical environmental problem. That typical problem is not caused by a few but by many polluters, has effects on many locations and has effects over a long period of time, and these effects can be modelled or indicated only with the greatest uncertainty. Climate change is the purest example. All firms and persons, without exception, are involved in causing it, effects occur in all countries on all continents, the emissions of CO_2 and CFCs exert their effects for centuries, and the range of predictions of effects is broad, in both a qualitative and quantitative sense. Even extended liability cannot sensibly play a role here.

Thus the application of even extended private liability remains primarily limited to the short term and mainly specifically assessed, local effects, caused by a small number of emitters, and affecting a small number of injured parties. With some hesitation, *extended private liability* is included in the list of macro instruments as the only structural one, from an unknown domain of other possibilities[1]. Its role in this study remains limited.

Macro cultural instruments: a choice

Environmental concern is widespread. Other things being equal, especially costs, a choice will be made by most producers and by most consumers for the alternative that is least environmentally harmful. Domains of choice may be very diverse. Buying one product or another, buying one brand or another as well as developing one product or another or working with one firm or another are various examples.

The experiences in the related field of health and safety may indicate the importance of such information. In the Seventies health and safety scores of larger chemical firms proved to differ substantially. The number of fatal accidents per man year or per dollar of added value differed between comparable firms by a factor of ten. Some firms got a bad name because of a number of publicized comparisons first in *Scientific American* and then other publications. Accidents leading to injuries and deaths directly imply costs. Improvements also cost money and management time. Hence from a short-term cost point of view leaving things the way they are might often be a financially reasonable policy.

[1] It is easy to put together some less macro, more incidental structural measures that still may have a broad beneficial influence. Here are a few: guaranteed amounts of money for environmental research at universities, also for policy oriented sciences; broad support for international consultation in scientific and policy circles through stable, well financed institutions; helping normalize policy oriented environmental effect analysis; breaking down barriers for environmentally beneficial technological improvements such as combined heat-and-electricity production. These instruments do not have an environmental target.

With the information on their operations becoming public, however, firms "below the line" experienced problems in keeping up worker morale, acquiring and keeping high quality personnel and keeping and developing market shares. In the long run any firm in that position would fail because high quality personnel and market shares would go to its competitors. Active in-firm policies for improvement were developed and became effective. Improvements are based on management policy aims, better procedures, and improved products and installations. This development was not the result of an information instrument, it occurred autonomously in society. If, however, an active instrument with similar effects could be framed in the field of environmental problems it could be very attractive.

How could publicly generated environmental information work in a similarly beneficial way? There are two developments on the society-environment interface towards environmental information systems. One, the environmental audit, takes the firm as the object of information. The other, the life cycle analysis of products, takes the product (or better: all processes required for the functioning of the product) as the object of information.

The *environmental audit* is an emerging private activity. The European Commission, however, is currently preparing a draft regulation on the "Eco-Audit" scheme, thus creating an instrument for environmental policy. At the level of collective private norms, organisations are formulating requirements on standards for environmental management systems, specifically ISO 9001 of the International Standards Organization and BS 7750 of the British Standards Organization. Many different types of information may be part of an environmental audit. Looking at the formally private BS 7750 of 1992, one can distinguish three potential elements, see Johnston (1993)[1]:
(1) the internal environmental policy of the firm
(2) the adequacy of procedures in relation to that internal policy
(3) the effects caused by the firm, both regular, including "cradle-to-grave"[2] effects, and incidental.
In current private audit practice two further elements are central:
(4) compliance with laws and regulations and with public aims
(5) the adequacy of procedures in relation to external public policies, mirroring 2, but for public instead of private policies.

The audit, as an instrument for environmental policy in the strict sense, would have to state its target in environmental terms, as in 3, possibly combined with parts of the compliance element, number 4. All other elements seem procedural in character, thus possibly belonging to instruments for environmental policy in the broad sense, that are not worked out here. Alternatively, these elements might be seen as part of a system of "enforced self-regulation", as part of an instrument otherwise lacking an implementation interface. See Braithwaite (1982) and Grabosky and Braithwaite (1986) on this subject. Elements 4 and 5, possibly backed up by elements 1 and 2, would form the working mechanism of interface I, while the actual performance would form the target in interface

[1] For a survey of the approaches to environmental auditing being developed by larger firms see UNEP 1990, especially the part by the International Chamber of Commerce.

[2] Interestingly, liability extends the responsibility of firms only towards the grave.

II, stating the effects on the environment. Although this picture does have its charms it is not precise enough for further treatment here. Most current laws and regulations do not state environmental targets but rather technologies and procedural obligations. To that extent the (procedural) "working mechanisms" of the audit would still not establish an instrument of environmental policy in the strict sense. Furthermore, these other targets have been embedded in other instrument mechanisms already, usually the prohibiting types. The audit then is a second line of enforcement strategy for some *other* instrument. An audit instrument could require a firm's environmental policies and procedures to be described in the audit before it may be called an environmental audit, in which case the target is still a procedure, not an environmental interference or problem. A final argument against an obligatory full audit (elements 1 to 5) as a cultural instrument is definitory one. The obligatory audit then is not conveying information on how procedures could be set up in the firm, but obligations on the procedures to be followed, as a prohibiting instrument for environmental policy in the broad sense.

How could an audit be set up as part of a macro cultural instrument for environmental policy in the strict sense? The framework as developed here would require a definition of object and target, as interface II, and a cultural working mechanism, such as interface I. Starting at interface II, I first restrict the object and target, *contrary to current practice but in line with the audit as required under BS 7750*, to the statement on the firms' actual effects on the environment, both directly and in the cradle-to-grave chains, i.e. element 3. This interface part has a strong similarity to the life cycle analysis of products. The objects differ, while the targets may be completely the same. The object of the audit could be analogously defined as "all processes required for the functioning of all products (including services) of the firm". For most firms the products are not final products, i.e. intended for consumer use. This poses severe practical and theoretical problems. Is the audit of a firm producing bulk polymers to include all actual applications and their "graves"? Is the audit of a petrol producer to include the production of the cars using the petrol? Is the audit of a car builder to include the production of petrol used in driving the car? For the moment, lacking data on specific products and methods to incorporate these data in a firm's audit, the cradle-to-grave part of the audit seems primarily applicable only in a qualitative screening sense. Since it is not yet clear whether a full audit will ever become possible, the full cradle-to-grave audit will henceforth be disregarded here. What remains at the other extreme of complexity are the data on the environmental effects of only a single firm, with problem-shifting to other firms as an attractive possibility. As a small step toward cradle-to-grave analysis this analysis might be extended to the effects of two or more directly related firms together, thus covering some responsibility elements in the chain. An practical example is the process that first produces chlorine, then vinyl monomer and then vinyl polymer.

There is one middle option between these two extremes, here termed the "cradle-to-product audit". It includes all processes of the firm itself and all upstream processes by other firms and also the processing of the wastes produced in these processes. It excludes final use and disposal of the products themselves. This solves some of the practical and theoretical problems of the cradle-to-grave audit but not all of them. Allocation problems in the usually diverging web of related firms, theoretically not yet developed and practically very complicated, will limit the application of this type of audit. Possibilities for problem shifting would remain limited, however, within these restrictions. Assumed in the further analysis here is that environmental information in the audit is restricted to this cradle-to-product type of analysis.

How can this "restricted" environmental information on firms be used? There seem to be two basic ways[1]. First, the environmental accomplishments of firms may be analyzed over the course of time. The information gathered should then take the form of an improvement analysis. Such an analysis indicates where improvements are possible and monitors whether improvement options are realized in fact, and if not, why not. The unit of one firm is then taken as a constant, to be compared over the course of time. This is the main function of the audit as seen by some practitioners in the field, see for example Cook (1991) and is consistent with the audit as required for BS 7750. The audit sums up the environmental effects, at least in terms of substances, as targets, of all single processes that together constitute the material operations required for the functioning of the firm. Generally, diverging scores on different emissions will make it difficult to state what is an improvement. Extending the target to problems will substantially increase the chances for an unequivocal result, as is the case with the life cycle analysis of products. Ranking and weighing problems, however, will often be required to assess whether a given change is an improvement or not. An authoritative set of weights would be most helpful here. Such a set could only exist if the set of problems is restricted to supralocal ones, treated in a non-site-specific manner. The weights in the audit should correspond entirely with those of the LCA. This seems a sensible approach. If the analysis would be restricted to only the operations of the firm problem-shifting, as by setting up dirty firms externally to the firm primarily analyzed, might still occur.

Secondly, the cradle-to-product audit information might also be used for comparing different firms[2]. Here the problem arises that firms differ in many respects and the unit "firm" is hardly relevant in itself. A car producer, for example, which has reduced its activities essentially to design, mounting and marketing will be clean compared to one that has integrated the whole production chain. Somehow, the absolute environmental effects, however measured, are to be compared using an entity that indicates the social importance of the firm and the type of its activities that cause the effects. Firms similar in size, with similar levels of vertical integration, and a similar product mix might be compared. This then is effectively the same type of comparison as that of one firm at different point in time. Such extreme similarities are seldom. A broader functioning of the comparison audit requires other comparison criteria. A first candidate is the cradle-to-product effects per unit of sales (value-added cumulated) as a correction on firm size. Next, "similar product type" or "similar production sector" would restrict the comparison to firms that could use the same technologies. The basis could be the amount of sales or the physical volume of production. Differences in specific products would not yet allow

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[1] If environmental effects were monetized, a firm's contribution to social welfare could perhaps be computed. As an experiment BSO, a Dutch based software house, made such a computation on its 1991 activities. The unsolved theoretical problems and the lack of information on environmental effects will keep this type of audit experimental for a long time to come. One awkward element in such computations is that the environmental effects caused outside the firm, towards cradle and towards grave, would also enter in the computations of all other firms involved. See critical comments on the BSO report by van der Vlies 1991.

[2] A quite senseless example is the lists of prime emitters of some substance that are regularly printed in US daylies, which suggest that the upper names are the bad guys. Size of firm, type of product, technologies chosen may be perfectly ligitimate reasons for being on top of the list for a specific substance.

the comparison of firms in terms of environmentally better or worse. Differentiation into several types of products could achieve grouping by equal societal function. Placing all other firms in the production column, including waste processors, would have solve the problem of vertical integration. In fact the firm is then no longer the unit of aggregation in the society-environment interface but its products or sum of its products involved. The analysis then broadens to something very similar to a partial life cycle analysis of products, excluding use and disposal. It is not yet clear if there is an independent place for such a comparative environmental audit in addition to the full life cycle analysis of the firm's products, treated below.

The improvement audit can be executed by in-firm or external private auditors. It is not yet an instrument of environmental policy. Audit standards, specifying methodology for a "good" audit, might be the macro cultural instrument. This *standard methodology for environmental improvement audits* is included in the list of macro instruments, but will not be worked out further. For simplicity's sake and for comparison with the life cycle analysis the target of the audit has until now been restricted to non-site-specific effects. A primary choice, however, still remains to be made on whether and how to include such highly concrete local environmental effects in an audit.

What would be the place of the obligatory disclosure of information by firms on the resource use and emissions directly by the processes of the firm only? To some extent this obligation already obtains in the US, creating a very limited sort of audit. The disadvantages of this type of interface have been indicated already. That obligatory audit, though very feasible, is not considered an independent instrument here, let alone a cultural one. It might be a worthwhile secondary instrument however, for various overlapping reasons. It might support extended liability as a preventive policy instrument. It might supply the data required for life cycle analysis and for the more extended cradle-to-product audit. It might also play a role in shifting the control activities on permits to private societal mechanisms[1].

The *life cycle analysis of products* or *LCA* is the basis for the main cultural instrument that will be used in the further analysis here. As was the case with audits, an analysis is not an instrument for environmental policy but may be a part of one. There are subtle but important differences in the ways it can be used for building cultural instruments at different levels of aggregation. At the lowest level of "macroness", the analysis might be applied by governments to specific products, accompanied by the publication of information. It might be applied to groups of products, accompanied by the publication of a comparison of these products, perhaps with the addition of an ideal as a target reference. Similarly, the comparison might be with the group average or group standard to indicate "substandard" and "above standard" products. Underlying the application for specific products is a whole range of induced private applications, to support favourable outcomes for one's product. These range from the most general strategies for a firm's market choice and product development, to specific requirements in the development of a product. If governments carry out LCAs on specific products, they will also indirectly set in motion some of these other applications.

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[1] See Braithwaite (1982) for a fuller treatment of "self-enforced regulation".

However, the LCA instrument might also be applied at a higher level of abstraction with a more macro character. Government might institutionalize a such a group comparison into a *system,* for example, for ecolabelling that differentiates between "good", "not good", and "bad." Or a more differentiating system for rating environmental performance might be developed. Such systems have been, or are being introduced by most Western countries and the EC, see Hartwell and Bergkamp (1992) for a description and discussion of the EC Council Regulation (1992) creating an EC eco-label. They usually incorporate the concept of life cycle analysis but not yet a worked out let lohe a definitive method.

Still one step higher in aggregation would be the authoritative standardization of LCA methodology (method plus tools). No specific case analysis by any government is involved. No specific action on further dissemination of information is added. No environmental comparison of products by governments is required. Only if someone illegitimately claims to make LCAs according to the standard, might there be a control function implied for government. This *standardized methodology for the execution of LCAs* is the most macro cultural instrument considered here. An institutionalized system for ecolabelling, one level lower, will not be considered separately. If a general standard exists, such systems might become institutionalized autonomously. On the other hand, a system for ecolabelling using this standardized methodology would support the broad use of LCA in society, as is the case with some structural instruments such as an obligatory LCA for certain decisions and claims.

The life cycle analysis will be worked out at the target level of environmental problems in Part Four. This means that a (functional unit of) product receives scores on the several environmental problems it contributes to. An overall comparison between products differing in many respects will usually require the ranking of the different product scores. Such a comparison will often be required, for example, in product design and supporting decisions on the purchase of products. This normative element could be part of the methodology provided by governments for the private functioning of the methodology in society, either through a procedure, i.e. a body that takes case-specific decisions, or through supplying a set of weights indicating the seriousness of all the environmental problems involved. In the latter case, the problem oriented life cycle analysis would become a "total problem analysis".

Macro financial instruments: a choice

Financial instruments have been defined here restrictively. They encompass taxes and deposit-refund systems only. Tradable permits are primarily permits, a prohibiting instrument, type 1. Paying for damages caused may be the financial result of a structural instrument, e.g. extended liability, but is not regarded as a policy instrument in itself.

Subsidies are also a financial instrument that directly changes prices. They fall off here as instruments for environmental policy in the strict sense, if they do not have an environmental target. An example are investments subsidies on certain technical installations. If subsidies would have emissions (or more aggregate units as a target such as problems) as their target, they in fact <u>are</u> emission *taxes, not subsidies.* Take for example an ideal subsidy to a firm that is related to its non-emission in the sense of emitting less than some stipulated amount. For each non-emitted unit the subsidy increases by a fixed amount. This would be the most ideal subsidy possible, avoiding unnecessary costs by not specifying any technology. This is equivalent to Mäler's (1974) description, "subsidies for the waste discharger in such a way that one more unit of waste discharge reduces the subsidy by $*q*." (p.8). However, simple analysis reveals that such an ideal subsidy is not an environmental subsidy at all. Imagine that a firm increases its emissions by one unit. For this increase the firm receives a lower subsidy, which situation is equivalent to paying the same unit as an emission tax. Such an ideal environmental subsidy then is a combination of a non-environmental lump sum subsidy and an environmental emission tax. The emission tax part is just a full emission tax, not a subsidy at all. The only difference between the environmental subsidy and the emission tax is the transfer of the non-environmental lump sum. The lump sum should be higher than the total amount paid in the emission tax component, since otherwise the sum total of both would not be a subsidy. This lump sum is not without indirect environmental importance. It has the effect that firms will stay in the industry longer and total output will be greater than without it, as compared to the single emission tax[1]. There seem to be no good reasons for such a lump sum transfer payment. Also, the basis for the transfer payment is not easy to establish practically. Being a lump sum, it should be independent of the firm's behaviour[2]. Hence, only historical performance can be a basis for differentiation between firms. New firms would then be excluded from the lump sum payment. The inclusion of new firms would make it worthwhile to split up firms, and receive the lump sum several times. In any case, as an instrument for environmental policy the lump sum is irrelevant by definition, even if added to an emission tax[3].

Thus the financial instruments remain limited to deposit-refund systems and taxes. First, the deposit-refund systems will be discussed.

Macro financial instruments 1: deposit-refund systems

Deposit systems again may have different targets. The deposit cannot be applied to problems directly, since the deposit is on physical units. As with the emission tax, an environmental problem may be broken down into its contributing substances. The deposit system might be applicable to each one. As noted above, there is one main difference between the deposit systems described by Bohm and Bohm and Russell, and the substance deposit, directed at all processes as developed here[4]. This is the macro level of the object of the substance deposit instrument as compared to their examples. With Bohm and Russell the deposit-refund system on chemicals functions as an indirect measurement of the level of one firm or a vertically related group of firms or group of consumers. The substance deposit system however has the full total of all economic processes in society as

[1] See Baumol and Oates 1988 for a full (but perhaps unnecessarily complicated) treatment and Sims 1981 for a survey of literature. Mäler 1974, p.8 states that if lump-sum transfers of any kind are assumed possible, "that bribes and charges are equivalent from a purely theoretical point of view". This would be true only for the allocative effects on the production level of the firms involved if the full lump sum would be taken away. Then, indeed, the bribe is not a bribe at all but only an effluent charge, here called an emission tax.

[2] Complications arise if the level of the subsidy depends on the behaviour of the firm. Most literature treats these complications. However, it then is no longer a fixed lump sum.

[3] This analysis does **not** show that subsidies, as an instrument for environmental policy sensu lato are an inferior instrument for reasons of efficiency in any instrument comparison. They are such only as compared to a full emission tax system, for example. If that is not available some types of subsidies might become first choice.

[4] See Huppes and Udo de Haes 1987, Huppes 1988, Huppes and Kagan 1989 and Huppes et al. 1992 for development over the course of time.

its object, that is the most aggregate object possible. The *substance deposit on total emissions* will be the main financial instrument in the cases in Part Five. The deposit system may also be framed as a "problem deposit", as the sum of all substances concerned. The *substance deposit on a total problem* then results. Both these options belong on the list of macro instruments. The substance flow analysis is the interface II part of the deposit system. A more elaborate description of it will be given in Part Four, as one of the two most aggregate level society-environment interfaces now available.

In its functioning the substance deposit is nearly equivalent to a uniformly applied emission tax. There are a few differences. Only reasonably competitive markets will transmit the price signal of the deposit system undisturbed. Net disturbances inevitably reduce to zero, however, since the total amount of a substance not flowing out is exactly what is paid for as emissions, so one person's gain is another's loss. There are not many situations where market failure leads to the collapse of the regulative functioning of the substance deposit. Even if at some point in the relations between processes there is no market at all, as for example when waste flows down the sewer to purification installations, the regulative function remains. The receiver then does not pay at all, and certainly pays no deposit. This implies that the user of the sewer has lost his deposit. The purification installations are thus not influenced by deposit payment. First, however, the deposit payment may still reward them indirectly. This is the case if purification can be done in a way that the substance in question is recycled, at the artificially high price level caused by the deposit. An example is the lead from lead covered roofs, extracted from the purification installations of the sewers where the acid-dissolved lead flows to. Secondly, deposit refunding may reward the owners of purification installations directly. This is the case if the installations effectively destroy or immobilize the substance. Hence even the entire lack of a market in the societal flow of the substance does not impair its regulative functioning.

There is another difference between the substance deposit and an emission tax, i.e. that of storable pollutants. With the emission tax, storage for later emission is attractive for delaying payment, for financial reasons[1]. The deposit does not stimulate storage. It stimulates acceptable disposal as fast as possible.

Macro financial instruments 2: tax systems
Taxes may be levied at emissions or problems as targets, and with either individual processes (and those of a single owner) or all processes, as objects. Both targets are feasible, if problems are formally defined in relation to emissions. The problem tax is then the collection of all attuned emission taxes on the substances related to the problem. The object "all processes" can not be realized literally, as there is no owner of "all processes". It can only be realized in the sense that the tax applies to each of them, individually, to the same degree, as a *uniform emission tax*, potentially transformed into a *uniform problem tax*. These are the principal macro tax systems.

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[1] There may be a loss of efficiency implied by the improper storage of certain levels of pollutants before their emission. See Lewis 1981, and Lee 1981 as cited by Lewis. This hardly seems relevant quantitatively.

Taxes with economic and not environmental interference-based targets, are instruments for environmental policy sensu latu only, if environmental instruments at all. By failing to uniformly influence all decisions relevant to an emission they are in principle less efficient. A tax on petrol for cars, for example, cannot cover all decisions relevant to the emissions from cars. The decisions covered are covered only very partially. Decisions on the design of cars and on driving style, for example, are only directed towards better fuel efficiency, not towards emission reduction. All other decisions that might reduce emissions remain unaffected. Such second-choice taxes may be relevant at the case level if the more strict financial instruments are not applicable. This is the case for example when real emission measurement, directly or indirectly as in the substance deposit, is not fully possible. The option often open is to estimate the expected emissions and make the *estimated emissions* the target of the tax. The estimate is not only less precise, it also loses the influence on emissions because of some behavioral parameter. Fuel consumption is a very crude way to estimate emissions. It could be improved. The estimate of NO_x emissions of a car, based on the number of kilometres driven and a test of the car under average conditions, is such an improvement. But it still omits the effect of driving style on emissions. All technical improvements are influenced by this tax. The level of the tax may be based on the estimated amount of the substance taxed. This *estimated emission tax* is also on the list of macro instruments. However, taking into account the effects of all substances on several problems is often the object of setting up "environment-related" taxes. Transforming these several emissions into their contribution to a single problem leads to the *estimated problem tax*. These two types of estimated taxes will be considered as a second-choice financial instruments, applied when substance deposit and emission tax are not applicable, and designed to approximate as much as possible these first-choice instruments. Environment-related taxes with economic, instead of environmental targets may be the third-best choice. Even if broadly applicable, they are disregarded here.

Applied together, the regulative financial instruments constitute a system of ecotaxes[1]. The comparison of financial instruments such as emission taxes and substance deposits, with other types of instruments involves one systematic difference. They create net proceeds for governments. This should be looked upon as an advantage rather than a problem. Given a certain level of public spending, current taxes with unwanted side effects are replaced by taxes with desirable environmental "side effects". See in this sense Terkla (1984) and Pen (1990). As with all the effects of instruments, only a complete model can give quantified results. No such models as yet exist however.

[1] Ecotaxes are seen by some as a general source of financing, replacing nearly all other taxes. The level of ecotaxes then is based on the need for proceeds, see in this sense WRR 1992 and von Weizsäcker 1992. Others consider ecotaxes primarily a regulating mechanism for environmental policy. Their level is based on aims in terms of emission reductions or acceptable emissions. Current taxes, then lowered to the amount of the ecotax proceeds, would supplement the ecostaxes to the level needed. Here ecotax is used in the second meaning, as a system of regulative economic instruments. It is not related to any specific outlays. Its proceeds are only complementary to other non-environmental taxes.

Macro prohibiting instruments: a choice
Prohibiting instruments 1: tradable rights
Tradable rights may be emission rights granted or sold by government, the original owner of the collective goods[1]. If trade is allowed between all owners of processes, the *generally tradable emission permit* results, as listed by Bohm and Russell, see table 2.4.6 above. This seems to be the most macro tradable right. It occupies quite the same level of aggregation as the uniform emission tax, both requiring the individual monitoring of each emission process but treating them equally as a group. The main difference is that in the one case government charges for the infringement of a public good, as a form of administrative liability, leaving the number of infringements to be determined by market forces. In the other case, the property rights on the public good are privatised, by restricting the total number of infringements on the public good that is allowable by private activities. The partly privatised right to the ownership of the public good may either be sold by government, as in the case of auctioning rights limited in total number or flow and duration, or the right to a certain flow may be "given away for free" to be henceforth a private property right. A given number of emission permits have no dynamic emission-reducing effects, but reduce costs dynamically. The equivalent emissions tax does both. The latter however puts no limits on structurally growing emissions, resulting for example from increased consumption.

Prohibiting instruments 2: direct interventions
In the economic literature the dominant instrument used in comparisons is the emissions standard for products and installations. However, such general standards have seldom been used in regulatory practice since actual emission measurement is not usually possible. It is only in a very few permits that the emissions allowable for a given substance are specified quantitatively and actually measured. In such circumstances an emission tax and tradable emission rights are applicable as well. The technical standards developed in individual permits are a most micro instrument, not relevant here[2] and disregarded further.

The *general emission standard* would be the most macro direct intervention instrument, with all substances separately serving as its target, and a group of homogeneous processes as its object. It is put on the list of macro instruments here, although the possibilities for actual implementation seem limited. Compared to the tradable emission permit its level of aggregation is much lower. Only products or installations of some specific type can be covered in one standard. In its implementation, the emission measurement of all substances specified is required, at all sites where the installation or product is used. This

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[1] If emission permits are sold by government at a fixed price in any amount desired, no trading will result. Nobody will pay more than what is to be paid to government and nobody will sell for less than what he has paid himself. In effect, a uniform emission tax will have been established.

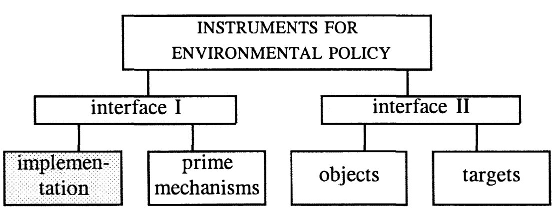
[2] The permit procedure decides how near the permit approach the ideal. In the Netherlands, Rijkswaterstaat, the body responsible for emissions into state owned bodies of water, has a procedure on individual permits that can increase static efficiency to a level equal to a uniform emission tax. In an internal and non-public procedure the marginal costs are estimated that would realize the amount of emission reduction aimed for. A number of emission-reducing techniques are modelled on this procedure. Next, in the permit negotiations with individual firms, the officials try to determine the cost levels of the firm, using the techniques they know as bench marks to test the reactions of the firms to other technical possibilities (personal communication van Luin, Rijkswaterstaat). Some mix-up of marginal and average costs may occur. Generally, the results are quite satisfactory from a static efficiency point of view. Most permit procedures specify techniques much more haphazardly.

is quite impossible generally. Reducing the application to one substance would make application more feasible but, correspondingly, less macro in character. The role of the general emission standard will remain limited even if measurement techniques improve.

Current regulatory practice, however, specifies a general standard not as an emission standard but as a general design standard. The general design standard may be based on some expected level of acceptable emissions that will result if the product or installation fulfils certain technical design specifications. If the quantification of emissions plays a dominant role in the definition procedure of the design standards, it could be termed an "*estimated emission standard*", analogous to the estimated emission tax. Though slightly less comprehensive than the general emission standard and nearing the border of instruments for environmental policy sensu lato, its practical applicability seems so much more realistic that this instrument is also put on the list of macro instruments. This standard is to be related to all substances emitted by the process. When the design is specified, the relative importance of the expected emissions of different substances would somehow have to be assessed in the procedure. This could first be done on the basis of their contribution to various environmental problems. The *estimated problem standard* would then result. If these problems, are next ranked, the "estimated total environment standard" would result, an option disregarded for the time being.

The possibility of standards based on the life cycle of a product, with problems as their targets, has been indicated above. It would be the estimated problem contribution not of one process but of all processes required for the functioning of the product. This life cycle-based *estimated product standard* has a more macro character than the standard estimated problems for one type of process. It is included in the list of macro instruments at a slightly higher level of aggregation than the one-process-only estimated emission standard for products and installations.

2.4.5 Administrative characteristics of macro instruments



Main approaches to administration

Which administrative characteristics of government are required for the implementation of the main macro-instruments distinguished above? Which characteristics would fit into the government-society interface? As situationists correctly point out, the way an instrument functions depends on the administrative circumstances in which it is applied. Governmental institutions and governmental culture are of prime importance for implementation, not just the societal circumstances within which an instrument functions. On the side of society, the prime working mechanism was included in the interface, as the main defining characteristic of this half of the instrument interface. The potentially relevant administrative characteristics are investigated here on the basis of main theories of how regulation functions or should function. It appears that very limited amounts of old-fashioned administrative means are themselves enough for the implementation of macro instruments.

The main difference in administrative characteristics relates to decision-making of officials. Bardach and Kagan (1982) name the difference "going by the book" versus "discretionary policies". They maintain that if rules are applied strictly, they often become unreasonable, while totally discretionary policies may also become unreasonable, for other reasons. A related distinction, involving styles of environmental regulation is that between formal and informal operations in environmental regulation, see Vogel (1986) on styles of environmental regulation in the USA and Great Britain and Hawkins 1984 for the basic terminology and the detailed description of Great Britain. Other comparisons with the seemingly exceptional style of regulation in the US are those with Sweden, see Kelman (1981). A similar comparison has been made by Covello and Kawamura (1988) regarding the US and Japan. The US approach is typified as being confrontational, with strong emphasis on rigorous scientific analysis and open adversarial procedures while the Japanese approach uses a cooperative model, with emphasis on negotiation and consensus building. What the Swedish, English, Dutch, and Japanese approach have in common is that such discretionary policies all require informal but authoritative operations by the officials involved. Kagan (1991) describes the US pathological adversarial legalism to be based on fragmented government with limited authority and party influenced decisions. He distinguishes four types of government, related to formal or informal, and hierarchical or party influenced modes of policy-making and conflict resolution (p.373):

Adversarial legalism	(formal & party influenced)
Bureaucratic rationality	(formal & hierarchical)
Negotiation/mediation	(informal & party influenced)
Expert/political judgement	(informal & hierarchical)

This basic distinction in formal and informal type of decision-making by officials goes back at least to Luhmann (1968). He distinguishes between *rule oriented behaviour*, or conditional programming, and *goal oriented behaviour*[1]. In rule oriented behaviour officials apply substantive rules, formal or not, indiscriminately, that is only differentiating according to the criteria stated in the rule. Their freedom is limited to interpreting rules, and advising on their adjustment. Goal oriented behaviour, on the other hand, leaves officials the freedom to decide on the means deemed appropriate for the goal they are supposed to further. When regulating technologies, environmental officials can hardly base their decisions entirely on rules. In environmental policy, some form of goal directed behaviour is dominant in Europe. General rules on products and technologies are the exception. Most policies are laid down in technical specifications in individual permits. The protection against regulatory unreasonableness is only given in procedural formal rules, that in a decision for example, "due account should be taken of all interests concerned". Bypassing the authority that makes the decision used to be exceptional. The number of cases of litigation now also is increasing in Europe however. Thus, procedural formalization is getting under way in there as well, influenced by EC procedural rules.

[1] There is a quite complicated connection, with Weber's *Wertrationalität* and *Zweckrationalität* on the one hand, and his formal and substantive rationality on the other.

Officials increasingly need procedural knowledge, on top of the ever-increasing amounts of specific substantive knowledge of processes and products.

In the USA substantive formalized rules have developed to a degree not yet imaginable in most European countries. Even so, substantive rules, whatever their detail, cannot usually apply to each case "automatically". Differences of opinion are solved through decision and litigation, at several levels. US procedural formalization has also reached levels not yet encountered in Europe.

Current developments in administrative sciences try to stop this quite endlessly and often senselessly spiralling of adversary moves. One way out is *horizontal government*, both as a theory on empirical development and a normative theory with an attractive potential for administrative development. Other names von Benda-Beckman and Hoekema (1987, p.3) give for this type of government are:

if one favours the development ◇ reflexive law, leading to participatory democracy,
if one is neutral ◇ neo-corporatist or situationist government
if one opposes the development ◇ interest-group democracy.

In neo-corporatist horizontal government officials develop and implement policies within the broad confinement of their domain in a dialogue with all those concerned. The dialogue concerns technical aspects, in the form of a communicative exchange, and also bargaining[1]. To prevent formal procedures by outsiders, formal administrative decisions are avoided as much as possible, either by not writing down decisions at all or by using non-regulatory, undefined forms such as private covenants. The sharp division between policy-making and implementation, controlled and safeguarded by democratic procedures on the one hand, and private responsibilities on the other becomes obscured[2].

The informal style of regulation as found by Vogel in Great Britain is perhaps not yet horizontal government. Contrary to horizontal government, the sharp distinction between "we" as regulators and "them" as regulatees has not yet been replaced by the "we" of the "we that are together responsible for improving our common environment". The British[3] situation as it is depicted by him and might still be, is the pure goal directed type of bureaucratic regulation, developed and implemented by a small, highly-educated, more or less impeccable administrative organization, in the finest civil service tradition. This style of government, highly attractive in its ideal form, is doomed if procedural formal rules lead to more and more control by higher ranking officials and courts[4]. The discretionary powers of the knowledgeable officials are lost in these procedures. The move towards formal, public procedures has even reached Great Britain, where litigation now no longer

[1] See Glasbergen (e.g. 1989) as a proponent of horizontal government; of "activating regulatory networks", Huppes 1989b for a critique on Glasbergen; and Hoekema (1990) for a critical analysis of "horizontal" developments.

[2] See for the US the paragraph on 'creative public-private partnerships' in Council on Environmental Quality 1991 and for the EC a document (EC Manual 1990) on a neo-corporatist approach to waste policy.

[3] In the Netherlands, policy development in many respects still resembles the British version described by Vogel. The head of the Amsterdam environmental control office once declared that the best permit is the one not yet issued. That situation allows a significant influence of the firm's behaviour, while the behaviour becomes fixed in a final permit. In Amsterdam, as in the Netherlands as a whole, the vast majority of all firms does not possess the legally required environmental permits.

[4] In Great Britain, juridical cases brought to court by officals, by those controlled, or by third parties, were seen as a failure in good administration. The number of cases is now soaring.

is the exception and the call for publicly controllable procedures is loud[1]. Extending the domain of environmental policy, as required by preventive policies, puts the "British" style of regulation under pressure, since maintaining a highly competent and motivated staff that is small is much easier than maintaining quality in a larger one.

Administrative requirements according to instrument
How do the macro instruments on the list relate to these different directions in public administration, as compared to micro instruments? When micro-instruments are used, the choice seems to be an unhappy one between the litigious US style of regulation and the neo-corporatist form becoming dominant as an alternative, and no environmental regulation at all.

Macro policy might be a way out of this unpleasant dilemma. The formal litigious style of regulation requires officials that are highly competent both technically and procedurally. Successful neo-corporatist administration requires officials that are technically competent and skilful in bargaining in complex social situations. Macro policy instruments, on the other hand, only require a very limited regulative capacity on the part of government.

Structural instruments
In the case of extended liability, legislation is required for its introduction, as with any new instrument. Extra court capacity is also permanently required. This, however, does not concern the environmental part of government. No claim on environmental bureaucracy is necessary, no choice of direction is required.

Cultural instruments
Developing and revising an authoritative system for the environmental life cycle analysis of products is a relatively simple affair. It requires a very limited number of primarily scientific personnel[2]. Research and development for the specification of environmental problems is another required activity. It would not take more personnel than are already assigned to such tasks in the environmental bureaucracy and the scientific community surrounding it. Authoritative advise on standards for ranking the different problem types would require an international body with national representatives and a small scientific staff. The officials required are high ranking, of limited numbers and need little specialized knowledge.

Financial instruments
For the substance deposit the Duty and Excise offices[3], a capable body in all industrialized countries, are the most apt organizations for primary implementation. The technical expertise required is very limited, comparable to the knowledge required for excises. Emission taxes require expert knowledge on measurement techniques. Estimated emission taxes require more expertise on environmental aspects of technology. Scarcely

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[1] An editorial in the British New Scientist on this subject (5 september 1992, p.3) makes a comparison between the procedures for regulating dioxins in the US and Great Britain. The British appraoch is unconditionally condemned.
[2] Maintaining an established system would not require much more than a few qualified officials per country, and a body of perhaps a few dozen persons in each country controlling updates of the methods and the general data base processes required.
[3] See the case studies on substances for more details.

any complex bargaining or additional rule making would be required. Adversary procedures on the environmental effects of technologies would be very limited, probably restricted mainly to waste handling. A limited high level scientific staff and personnel for routine measurement and taxing are required. Litigation would be rare.

Prohibiting instruments
Tradable permits require the same type of personnel as emissions taxes. Design standards require a more intricate knowledge of technical developments and their economic aspects in all industries concerned, than all other macro instruments. In this bureaucratic respect they belong to, and are similar to current micro policy instruments.

A choice on style of regulation?
By moving in the direction of macro instruments it seems that the unpleasant choice can be avoided between a formal litigious style of democratically controlled administration and a democratically uncontrollable horizontal style of administration. Structural instruments do not involve environmental officials in actual decisions. Hence horizontal government makes no sense there. No formal public decision-making procedures are required on a case basis, so there are no litigious procedures either. Litigation does occur of course when liability suits are brought to court. That litigation does not concern public policy however. The main cultural instrument, the authoritative availability of LCA methodology, does not involve any public officials making decisions at a case level either. Financial instruments require more administrative activities. These are of a highly technical nature, very much comparable to the payment and exemptions from payment associated with current excises as on alcohol. Duties and excises are one part of public administration where a neo-corporatist approach is not advocated and litigation is relatively scarce. The number of tax-like decisions is higher with the emission tax than with the deposit system. Tradable emission rights, in a not overly differentiated system, are comparable to general emission taxes. With the adoption of the estimated emission tax, the burden of discretionary decision-making increases somewhat, with larger numbers of pressure groups becoming involved and litigious decisions increasing, for example on the demarcation of products in or out of the group being taxed.

One significant step towards increasing the numbers of discretionary decisions is made in the adoption of general product and installation standards and estimated product and installation standards. In those cases choices are necessarily arbitrary. Which products belong to the groups specified? Why is the level of allowable emissions set at that specific amount? Such decisions may be a matter of life and death to the specific firms involved, with either litigious procedures evolving, or with a "horizontal" smoothing of the political process.

An example of the types of decisions involved in estimated taxes and measured or estimated standards are transport emissions. A tax on estimated emissions could be set for "internal combustion engines used in transport", according to several of their expected emissions. This group would include diesel and petrol engines but would, e.g., not include gas turbines, steam engines, Sterling motors and electric motors for vehicles. The electric car might use the most energy in terms of a life cycle analysis, in its requirement of coal-fired power stations with high levels of acidifying and global-warming emissions, but it would not be taxed. Emission taxes and standards seemingly avoiding this choice, based for example on estimated life cycle emissions per kWh of power delivered, could easily be drawn up so that only low compression petrol engines with denox catalytic converters can meet them.

A further major step brings us where we are now, to the direct regulative policies covering more and more individual private decisions. In that situation a choice between only the formal-litigious style of regulation and the horizontal one seems unavoidable.

Economy-environment trade-off

Successful environmental policy[1] sacrifices a certain amount of production and hence consumption, in market terms, to realize environmental improvements. The amount of consumption sacrificed for a unit of environmental improvement should remain within reasonable limits. There now seems to be an interesting difference between the main instruments distinguished in determining the trade-off between costs and environmental benefits. That trade-off fixes both the cost-effectiveness or efficiency of instruments and their effectiveness in terms of environmental results.

With structural and cultural instruments decisions on the trade-off are taken on privately, in society. The instrument helps to specify the half required for the trade-off, the environmental component. The other half, the costs incurred in preventing or diminishing the environmental effect concerned, are not part of the instrument and not even known to any environmental agency implementing the instrument[2]. The same applies to the tradable rights for the version in which trading in emission rights is purely private.
With financial instruments it is only the trade-off that is authoritatively fixed, all other decisions are left to private individuals and organizations. The trade-off is the kernel of the interface. This also is the case with tradable emission permits such as prohibiting instruments.
It is only with direct interventions, the second type of prohibiting instrument, that the trade-off is decided on, usually implicitly if at all, by the bureaucracy, with the results of that decision becoming incorporated in the instrument.

Thus, moving from direct interventions to other types of macro instruments makes superfluous the technical and economic knowledge in the bureaucracy that was formerly required for decisions on policy measures. All financial instruments require that one generally valid public decision explicitly be made by government on the level of the trade-off. With tradable rights the decision is also made, but only implicitly. With structural and cultural instruments, government takes no decision on the trade-off but leaves the trade-off to private society.

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[1] Unsuccessful environmental policies involve costs without results. Results 'for free' or even better, technological progress combined with environmental improvements, will result very seldom from public policy. If private advantages are there, no instruments are needed. Such improvements are very common among firms. See the 'Pollution Prevention Pays' programmes in the US and Europe, e.g. Huisingh (1986) and Berkel (1990).

[2] The preferences involving alternative choices depend on many factors, including environmental effects, as well as differences in several other respects such as functions. There is then no specific trade-off defined since the costs do not enter the decision as a seperate entity.

2.4.6 Conclusions

The conveniently systematic definition of *objects* and *targets* that constitute the society-environment interface cannot be repeated for the government-society interface. The social side of the government-society interface is defined in terms of the *main mechanism influencing addressees*. The governmental side is loosely indicated as *a certain mechanism of implementation*. The definition of macro as well as micro instruments in terms of these two interfaces is specific enough for a general analysis.

The division of the social working mechanisms into the four main types chosen, prohibiting, financial, cultural and structural instruments, has proved robust when confronted with classifications of policy instruments for economic policy. The main types of instruments distinguished in this way may be translated into the four types of social working mechanism, each at higher and lower levels of aggregation.

The primary classification resulting has been confronted by instrument literature from several disciplines. Two disciplines deserve special mention. Administrative theory seem to be biased against macro instruments. The instruments for environmental policy distinguished by economists fit into the scheme developed, if target and object are further specified. The harvest of macro instruments was small however. Only some prohibiting and financial instruments exceeded the micro level of individual processes or types of processes.

TABLE 2.4.7 MACRO INSTRUMENTS PRELIMINARY CHOSEN

Structural instruments	◇	Extended liability
	◇	(Possibly later: obligatory LCAs for some private or public decisions)
Cultural instruments	◇	Standard methodology for problem oriented life cycle analysis of products or functions
	◇	Standard methodology for improvement eco-audit
Financial instruments:		
1. deposit-refund systems	◇	Substance deposit on total emission / one problem
2. taxing systems	◇	Uniform emission tax / problem tax
	◇	Estimated emission tax / problem tax
Prohibiting instruments:		
1: tradable rights	◇	Tradable (auctioned) emission permit / problem permit
2. direct interventions	◇	General emissions standard / problems standard per type of product or installation
	◇	Estimated product standard, on life cycle-based problems
	◇	Estimated emissions standard / problems standard per type of product or installation

The further design of macro instruments results in a list of eleven instruments, see table 2.4.7 above. Two main instruments added will be worked out further in Part Four:

◇ The *methodology for the life cycle analysis of products, with problems as targets*, as the most aggregate cultural instrument,
◇ the *substance deposit on total emissions of one kind or on all emissions related to one problem*
as the most aggregate financial instrument.

The higher level structural, cultural, and financial instruments may avoid the unpleasant choice between current tendencies towards either exploding numbers of formalized policy with widespread litigation, or horizontal policies that effectively lack democratic control.

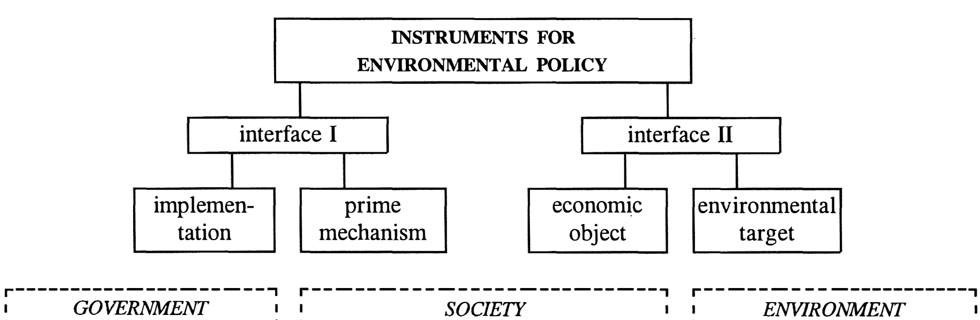
2.5 CONCLUSIONS ON THE GENERAL FRAMEWORK

For the design and analysis of policy instruments a model is required that links the political decision on their implementation to their expected environmental effects. Such a model does not exist. An approach has been developed that splits up the effect chain into three (sub)models, connected by two interfaces. The range of the total model is restricted to the chain from political decision to the desired effect in the environment. Effects of environmental changes, in the form of feedback loops to government and society, fall outside this range. Nor is a feedback loop from society to government considered here. Thus the main model has a simple uni-directional causal structure. This single string, triple model of government, society and environment has been specified only in some main submodules. This instrument analysis has been structured in terms of the interfaces between government and society and society and environment.

The design domain has been limited to instruments for supra-local problems, analysed in a non-site-specific manner. Purely local problems are disregarded. The local, site-specific effects remain the domain of individual permits and zoning laws. If aggregate policies become more effective, the number and stringency of locally required instruments will shrink but will certainly not disappear.

Instrument analysis not only can been aided by the models and interface distinctions. Instruments can even be defined in terms of the two interfaces between the three parts of the main model, as the symbolic interface between government and society and as the material interface between society and environment. The instrument definition is thus primarily independent from the partly disputed, partly non-existent models of society and the environment.

At the symbolic interface, instruments are defined in terms of the *prime social working mechanism*, set in motion with addressees of the instrument, and of the type of *implementation* required. Four main categories of social mechanism are distinguished, corresponding to the four main types of instruments: prohibiting, economic, cultural, and structural, in a roughly increasing order of potential aggregation. There is no neat classification of basic forms of implementation.



At the material interface, the *object* and the *target* fully define the interface. The objects are processes and groupings of processes, of which thirteen categories have been distinguished, roughly at an increasing level of aggregation. The targets are types of interference and their aggregates in the environment. The most fruitful road to aggregating targets seems to express the effects of interferences in terms of their contributions to a limited number of environmental problems. Interferences have been restricted practically to the extraction of substances from the environment and the emission of substances into the environment, excluding for the time being other types of interference in the environment. Ten categories of target, roughly at an increasing level of aggregation, may be distinguished. Feasible combinations of an object and a target as aggregate as possible have been investigated, as the interface II part of possible macro instruments.

Two types of society-environment interface seem particularly promising elements for macro instrument development. The first is the environmental life cycle of a product, functioning as the object, combined with its contributions to all environmental problems, functioning as the target. The problem oriented environmental life cycle analysis of products (LCA) fully specifies this interface. The other is the contribution of all economic processes, the object, to the emissions of a given substance, the target, or to one problem functioning as the target. The substance flow analysis (SFA) of the economy fully specifies this second promising interface.

Following a survey of the instruments in several disciplines, and the definition of several new examples, a list of eleven main macro instruments has been drawn up, see table at the end of the preceding chapter. The two main new additions to this list are a cultural instrument, the *authoritative methodology for problem oriented environmental life cycle analysis of products* incorporating LCA at interface II, and an economic instrument, *substance deposit on all societal flows of a substance* as an indirect method of broadly taxing emissions, based on SFA at interface II.

PART 3 PRINCIPLES
PRINCIPLES AND STRATEGY FOR POLICY DESIGN

3.1 INTRODUCTION TO PRINCIPLES AND STRATEGY

The aim of this Part Three is to construct a strategy for instrument application that conforms to a number of principles that are as widely accepted as possible. The type of principles considered here are quite general, encompassing such ideas as individual liberty, equality, justice, security and prosperity. In order to arrive at this aim a number of steps are required. The first question discussed, in chapter 3.2, is what the role of such principles should be. Can they be applied to instruments in general or should decisions on instruments be based on the specific situations to which they apply? The position developed is that *principles guided choice*, as opposed to rigid consequentialism, is the most reasonable approach. Having clarified the role of principles, the next question is which are the principles relevant to assessing instruments for environmental policy, handled in chapter 3.3. A survey of general principles, a description of their application in environmental policy, and a selection of them are the three main elements of this chapter. Having selected a number of principles, the final work to be done, in chapter 3.4, is to use them as the basis of a normative classification of instruments and construct a strategy for instrument application, the *flexible response strategy*. The concluding chapter 3.5 summarizes the results.

3.2 PRINCIPLE-GUIDED CHOICE

3.2.1 Introduction

Development and choice of instruments and policies for environmental protection can be structured in procedures. Democratic procedures, for example, will yield choices that may be quite different from those in autocratic government. In that sense procedures decide instruments and policies, not principles. For democracy to develop and sustain itself effectively in procedures, principles would be required at the collective level. These principles would select policies only indirectly through the democratic procedures based on them. Ultimately, as far as material choices are involved, only individual preferences would be decisive. This position would make a normative stand on instruments and policies at a societal level superfluous. Only the expected empirical outcomes of the policies are required for a choice. Those making the decision are the same persons as those that will be confronted with the outcomes. They have their preferences on these outcomes, they make up their minds, and the, democratic, collective decision procedure, produces the choice. Implementation then will usually move society into the empirical states expected. In welfare-theoretical terms, the social welfare function builds *only* on individual preferences. This position, itself inhomogeneous, is that of *consequentialism*, *pragmatism*, or *extreme utilitarianism*.

Other welfare functions are possible, as was first formulated by Bergson (1937, and quoted by Braybrooke and Lindblom 1970), that also incorporate purely collective terms, as aims to be realised even if costs are involved. Ethical positions on income distribution are a primary example from the domain of economic policy. Such positions might still be utilitarian, based on an empirical relation of income distribution with welfare, as in the maximin utility principle[1]. The position would still then be that of a consequentialist. The position may also be based on a collective ethical principle, that all men are equal, at least in some respects and to a certain measure, irrespective of any empirical relation with individual welfare. Lacking an established term for this position I here name it *ethicalism*. In ethicalism, also purely collective normative principles, not based on individual welfare, play an independent role in deciding what is good and bad, or better and worse.

There also is an intermediate position, in which at least some normative principles are (partly) based on expectations of their outcomes. The ethical position on income distribution, for example, might at least partly be a derived one, based on an assumed empirical relation between income distribution and the total, individual-based, welfare of society. Similarly, collective decision-making rules, like democratic ones, may be based on the assumption that they will further the individual utility-based welfare. This intermediate position differs from consequentialism in that a choice in some situation is made *without the consequences of the choice in that particular case being known*. It differs from ethicalism in that utilitarian reasons for social choices are fully acceptable. That intermediate position has been named the *principle-guided choice* position here.[2]

The evaluation of instruments and policies, in the latter two approaches, requires substantive principles. These principles are the normative criteria stating what is "good" and what is "bad." Such principles have a normative, component and an empirical component specifying the reality to which the norm applies. The aim of this chapter is to discuss the role of specific principles in the analysis and design of instruments for environmental policy. The principle-guided choice seems the relevant approach here.

These three main approaches to the role of principles are connected to another typification of decision-making. If consequentialism is practically impossible, as already maintained in Part Two, an alternative position might also be that of incrementalism, see Braybrooke and Lindblom (1990). If big changes have unpredictable effects, public policy should limit itself to changes in small steps and incorporate feedback on their results. The domain of policy is then limited to the domain where reliable knowledge of empirical effects is available. In principle-guided choice, and all the more so in ethicalism, such limits on the domain of policies do not obtain. Decisions on constitutions may be based on their assumed but difficult-to-prove conduciveness to welfare, as Beard (1913) argues. Similarly, war might be declared for purely ethical reasons. Such extreme changes are not considered here. Only changes in the instruments for environmental policy are considered. These may be smaller or greater. Even greater changes can be introduced gradually, in increments. Without an overall strategy, on which any single decision has to be based, such incremental steps could never evolve into a coherent whole. That strategy is to be developed and chosen is the position taken here. It is the main task of this study.

A related position on decision-making is that of satisfying behaviour, both for individuals and collectives, see Cyert and March (1963). The decision-maker is confronted with costs

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[1] See for example Barry 1989, pp.81-3. This maximin utility principle is implied in Rawls' difference principle that states that only those inequalitites may be just that are advantageous to the poorest.

[2] A similar position is taken by Buchanan 1985, p.35ff, for the applicability of the market system, relating the role of efficiency and ethics in argumentation.

when extending the number of behavioral alternatives. These costs relate to producing clear information on the consequences of an alternative action and to the increased complexity of decision-making if the numbers of alternatives grow. The rational decision-maker will thus limit the number of alternatives, with some form of incrementalism emerging. Again, for principle-guided choices such limits on the domain of policies do not exist. Choices can be made according to certain principles regardless of knowledge of their effects in specific situations. However, such principle-based choices might ultimately be based on general notions of outcomes that can usually be expected.

3.2.2 Against rigid consequentialism[1]

Rigid consequentialism, pragmatism, and extreme utilitarianism are viewed here from their common aspect that it only is the results of actions that count. Extreme utilitarianism (Hayek's term, his 1960, p.159), as first advocated for politicians by Macchiavelli and for behaviour in general by Bentham, Hume and Mill (see Hayek 1960, pp.158-9 on the subject and for references), sees choice of action based only on the expected results or practical consequences of that action. The rational actor model summarizes the approach: (probabilities on) the consequences of actions should decide their choice. That position has been implied in several disciplinary approaches encountered in previous section. It is most explicit in the dominant situationist approach in administrative theory, as exemplified there by Bressers. Every choice of instruments should depend on the circumstances in which the instruments are to be applied. Only these circumstances can tell which consequences are to be expected. A choice between instruments then can be made on the basis of comparing outcomes. Since situations differ, no general statements on instruments seem possible. The mainstream of economic analysis surveyed by Bohm and Russell also seems to belong to this approach, if not in fact, at least in intention. This analysis of instruments is primarily directed at a description of different situations in empirical models, where a given instrument has more attractive outcomes and should thus be chosen. Models as yet without quantification of place and time, also used by economists, might be seen as a step towards the full empirical specification that belongs to the extreme utilitarian approach. Alternatively, they might be seen as reasoned selections of certain instrument types, the instruments serving as "principles", in the manner of principle-guided choice. Adherence to the tradable emission permits ideal, quite common among economists, might constitute such a position. To be acceptable however, the principled reasoning should trace back the arguments to more fundamental principles, giving general arguments for the choice of such an instrument. Fundamental principles, hopefuly, are few in number and broadly accepted.

Hayek opposes extreme utilitarianism, on several grounds. First, it requires full knowledge of the consequences, which never is available. That practical difficulty cannot be solved by retrospectively (ex post) seeing how an instrument has worked, as incrementalists for example maintain. It is only the instrument concerned that has been applied in some specific situation, not its alternative. The alternatives would still have to be compared by means of a model for the "ex post prediction" of their results. If that model would incorporate all the particulars of the situation, the results would be indicative for that historic situation only. If not, the comparison is crippled. No such

[1] The title owes to Grisez' 'Against Consequentialism' of 1978.

models, case-specific or otherwise, are effectively available for a comparison that includes structural, cultural and financial changes in society[1]. The principle-based choice of instruments, if developed, could therefore easily be more practical.

A second, more fundamental reason Hayek gives is that societies function as they do because choices on principles have been made already. Western societies have been moulded, for example, according to the principles of the Enlightenment, freedom and equality, that are actively applied in every social activity, also in more practical aspects of social organisation, as in a sharp distinction between a public and a private domain and a division of powers to safeguard freedom and liberty and to limit inequalities[2]. Empirical relations are what they are because of these past choices. Any choice of instruments may be in line with these principles or it may go against them, regardless of their specific effects and the relative attractiveness of these expected effects. The choice of policy instruments as societal steering mechanisms, can hardly be neutral in these fundamental respects. Instruments will always differ in the degree with which they fit into the existing normative superstructure, e.g., by restricting liberty more, or less and creating equaity, more, or less. Relating policy choices, whatever their specific effects, to existing principles is therefore required if society is to remain an ordered, i.e. principle ordered, unity. This might not be true for one individual decision on an instrument with a limited domain. It certainly is true for the strategy that guides the instrument choices in all decisions in all domains of policy.

Finally, there are some difficulties associated with extreme utilitarianism that are specific to the field of environmental problems. First, there is the possibility of environmental effects lasting a very long time. CFCs emitted now will contribute to global warming and ozone layer depletion for several centuries. Depletion of fossil fuels (see Part Five), and probably of minerals as well, will become a serious problem in several thousand years. Effects on future generations can hardly count in an extreme utilitarian approach[3]. Secondly, the collective nature of the goods involved makes it hard to specify what the contribution to someone's utility of one specific policy measure will be, even if effects were fully known. How can one assess the effect one's individual utility of a diminished global biodiversity?

To some, the difficult and impractical abstract reasoning about principles for policies will seem far removed from the reality of policy making. In a comparison of the principle-guided approach to its alternative, extreme utilitarianism, it is the latter however that seems more impractical. Any attempt to evaluate different types of instruments only in

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[1] Because of these limits on available models, ex post policy evaluations effectively are limited to the most controlled part of the instrument, its implementation, or in terms of Marcus 1980, the government outputs as contrasted with the wider outcomes. If things go wrong at the output level, it is very probable that the intended effect will not occur either. If implementation is successful, that is in output terms, one still does not know the intended and unintended effects through all relevant societal mechanisms. The full outcomes are not usually researched by policy analysts as Marcus, p.11, states.

[2] The sharp division between the public and the private domain that developed in the Eighteenth century has had quite material consequences. After the French revolution, all shop signs extending from private houses over the public street, were removed.

[3] Technically it could, by extending the model of effects to that long-term future and by not discounting future negative benefits. Putting the discounting rate at zero has odd effects, like making almost any investment project worthwhile.

terms of their predicted consequences will be hampered by a lack of the complicated models needed to specify these effects. They will not become available in the near future either. That seemingly down-to-earth approach simply cannot work practically. Extreme utilitarianism might be *forced* to work, but only at a high price. Practicality might be forced, either by reducing instrument alternatives to a narrow scope, to incrementalism for example, or by limiting their application to a narrow domain, e.g. the current one. Only in that way can practical choices be founded on predicted effects. In the former case the choice is made beforehand, unfounded. In the latter case it depends on the way the model has simplified reality, and is also essentially unfounded.

Very similar to the distinction between rigid utilitarianism and principle-guided choice is that between planning as a goal-directed activity and law making as stating a set of rules that disregard to a large extent the specific outcomes of the related actions. The now very general acceptance in the Western world of the outcomes of actions as the criteria for their assessment has brought along a general shift in policy towards planning that, in an increasingly complex society, becomes ever more encompassing and complex. Environmental policy is one clear example of this tendency. The position of principle-guided choice advocated here would imply a shift back from planning towards legislation[1]. Of course, laws and rules are not determined in complete disregard of expected outcomes.

3.2.3 Principle-guided choice

Principle-guided choice as defined and advocated here for environmental policy is the only option that can avoid the implicit conflict with the existing general principles that shape empirical reality in society. It takes these existing general principles as the basis for instrument evaluation in the domain of environmental policy as well. With these principles, instruments are evaluated in disregard of the specific situations in which they are to be applied and of the generally unknown effects in these specific situations. Of course, the specific formulation of principles will be disputed and will develop in the course of time. These principles do not exclude reasoning on empirical reality. They even have been, and will be developed also in consideration of such general empirical reasoning. Ideally this reasoning would take the form of explicit general models, that abstract from the particulars of specific situations. Principles not only apply to situations as have been modelled, they may even guide the development of models. The sharp division, for example, between government and society in the main model structure here, is based on the principle that a clear division of responsibilities between public and private domains should remain. It is not necessary that reality fully behaves accordingly, as reality now shows. Such developments in reality, however, should not be contributed to if they are undesirable. The model structure developed here is therefore not to be seen as the framework for detailed empirical models, as yet incomplete, that are required before a policy choice can be made. It is rather a frame of reference for the application of principles to the selection of policy instruments. That selection of instruments is made on the basis of principles, before the particulars of the situations of their application are

[1] The position developed in this chapter builds on Huppes-Cluysenaer 1984. There, the arguments against planning in terms of an empirical science-based activity relate to the epistemological nature of empirical knowledge especially of the social sciences. All pervasive planning undermines social science, converting it to science fiction.

known, if indeed they ever will be. Of course, such a general model can also play a role in case-specific modelling.

A mild version of the principle-guided choice approach is adhered to here. Preliminary choices on instruments and their application may be made in advance, based on general, not case-specific, modelling. Such choices may then result in an instrument strategy for environmental policy, as is the case here. The practical applicability of the instruments chosen may, however, differ from situation to situation. Hence there remains an empirical level of analysis that further decides how the instrument strategy may be developed into practical policies. At that empirical level, however defined, the full choice of all possible instruments is not open again. The case gives empirical evidence on the applicability of only a few different instruments and it allows the quantification of the instruments chosen. The preferred order of instruments remains fixed, through their reasoned relation to principles in the "general case," whatever their predicted scores in specific cases.

Principle-guided choice might imply "big changes" as contrasted to incremental changes. Braybrooke and Lindblom (1970) thus characterize policies according to the extent of change involved as well as the extent their consequences are understood, (p.67) a relevant variable here as well. According to them, the general situation, is that of small changes with a low level of understanding. Big changes with a high level of understanding do not exist, while big changes with a low level of understanding are a risk that should not be taken in more or less satisfactory situations. Small changes with high understanding occur, but are seldom. Making a virtue of necessity they propose directing policy analysis at marginal changes, at *disjointed incrementalism* as the best way to analyze policy (Ch.5). In the related design of policies, those options are most relevant that may be analyzed as additional and are not excessively interwoven with many other variables. Contrary to expectations, the mode of analysis they advocate is very similar to the one advocated here, i.e. to use very general notions on empirical relations, not to try to predict the specific consequences of specific policies. Macro environmental policy, especially in terms of structural and cultural changes, only seemingly necessitates fundamental change. This is not the case. The introduction of structural instruments, e.g., may take the form of adjusting one element in liability rules now and another one in five years time. The only large-scale change involved is that the changes point in one general direction, and are cumulative in that sense. Thus, not only does this study share their view on the inaptness of the synoptic ideal, as they convincingly depict it (pp.50-54), but it also agrees to the need they express to advance in smaller policy steps. There is one main difference with their approach however. The strategy of disjointed incrementalism reverses the ends-means sequence and advocates the setting of objectives in relation to the means available (p.93). The difference is mainly one of psychology however, in terms of emphasis. They also see policy development as a principle-based activity, with general notions and strategies that help guide the development of practical policies. Their subject is how such practical alternatives can become viable politically. The subject here is to formulate the strategy for macro-environmental policy, thus constructing the setting for practical policy development in terms of a coherent set of alternatives for the long run. They concentrate on how such alternatives may be specified in a way that allows them to become practical policy. Though not treated in this study, the question of how to implement strategies practically, in phased alternatives, remains one of primordial importance, and there is much to be learned here from Braybrooke and Lindblom.

3.2.4 Conclusions

The principle-guided choice, a mild version of it, is the approach followed here for instrument choice. In that version the most preferable instrument is chosen as long as it can be implemented. An emission tax or tradable emission permit for diffuse emissions by mobile sources is clearly not applicable and cannot be chosen sensibly in such a situation.

The principle-guided choice seems the only means to prevent or at least minimize adverse structural changes in society that may help undermine the social order.

The principle-guided choice seems the only non-arbitrary approach to instrument choice that is practically feasible.

Actual implementation in a series of politically viable alternatives, the subject of policy analysts, although not treated here, is of paramount importance. The principle-guided choice advocated in this study is about how a background for the coherent construction of such short term alternatives may be developed.

3.3 PRINCIPLES FOR ENVIRONMENTAL POLICY INSTRUMENTS[1]

3.3.1 Introduction

How may the principle-guided choices become operational? A first requirement is the definition of the relevant principles. That is the task set for this chapter. Principles may be formulated at several levels. In the environmental domain, 'sustainable development' is a high level principle[2], encompassing broad societal and environmental elements. It requires specification before it has enough empirical content to guide choices on policy instruments. That specification has a normative definitory element and an empirical element. In the Brundtland report, for example, a definitory choice was made to include the attainment of 'minimum living standards' as a constituent element of sustainable development. That part of the definitory specification is purely normative. The specification also may be based on assumed empirical relations, thus leading to derived principles. Suppose a certain amount of global warming is deemed "non-sustainable". In that case the emission levels of all global warming substances together should not exceed a certain level, or too much global warming would occur and development would not be sustainable. Moving from high to low level principles may be a stepwise procedure. At each step there are further definitory choices, that narrow down the sub-domains of the principle, and further empirical relations, within each specified domain[3]. A further specification of the allowable emission levels for each global warming substance could constitute such a next step. Sustainable development functions as an umbrella for a number of quite independent subprinciples. When developing principles as to content, it seems a wise approach to first specify the independent elements before subsuming them under the umbrella principle. A first main choice here is to distinguish those principles which apply to the environment from those applying to government and society.

Environmental principles were implicitly treated in the discussion of environmental targets in the preceding part. There, environmental quality was the primary, but as yet quite empty principle. It may be defined, as is the case there, in terms of health risks, quality of nature, and the functions of the environment in production and consumption. This is purely normative specification. Then a certain non-specified model was assumed to arrive at 'environmental problems', as the derived principles. That non-specified model has two elements, one empirical with ideas on how acidification, global warming, etc. affect the value areas concerned, and one normative, indicating the relative importance of health risks as compared to a certain consumptive function and to the quality of nature. Since that model is not specified here (it has never yet been specified) such a derivation of 'problems' from the three main value areas is a purely subjective normative affair. Putting it the other way round, the quantified ranking of environmental problems into "the total problem", is the primarily definitory specification of "environmental quality". Lacking empirical specification, such a ranking is purely normative by nature. The next

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1 This chapter benefitted from the comments of R.J. van der Veen.

2 *Principle* is used here more or less interchangeably with *value* or *norm*. As the empirical content or specificity goes up, a principle increasingly becomes an empirical quality. Norms as quantitative standards are the most 'empirical' values or principles applied in a regulative context.

3 We explored this theme of levels of principled reasoning in a study for the Department of Public Works (Rijkswaterstaat), see van der Voet et al. 1990.

step in principle specification, translating problem contributions into substance emissions, for example, is mainly empirical.

The empirical models used in constructing derived principles may be highly formalized for a small part of the evaluation chain, as in the example of specifying allowable emission levels per substance[1] if an allowable level of problem causation has been fixed. The models relating substances to problems are not usually well defined, being not much more than a conceptual structure with some notions of main relations. Potential health problems due to emissions of toxic substances constitute such an area of not-yet developed empirical modelling. The distinction from the normative definitory specification is not so sharp then. For some environmental problem types models have been developed and tested against some empirical evidence. The two main examples are global warming and ozone depletion.

For societal principles a similar structure of ultimate and derived principles is possible, "ultimate" always being relative to a still more basic discussion of the nature of the values involved. The values in society may be split into two classes, individual values, such as personal self-expression, and those on a collective level, relating to how society should function, e.g. the principle of procedural equality. It is the latter type of principles that are relevant here for establishing the relative attractiveness of policy instruments.

The questions to be answered in this chapter can now be posed more precisely:

◇ what are the principles for society in general against which institutional arrangements, including instruments, are to be measured? (section 3.3.2)
◇ which of these societal principles and in what form, are relevant for establishing the relative attractiveness of instruments for environmental policy in particular? (section 3.3.3).

3.3.2 General principles for societal institutions

What then, are the basic starting principles? For widely differing types of instruments, ranging from liability for environmental damages, to a methodology for life cycle analysis of products and to tradable emission permits, a high level of abstraction is required, to cover them all under the same principles. These instruments, through their multiple application, may even influence the nature of society, in terms of its basic institutions. In instrument choice the highest principles for societal organization may thus be involved. The general background for evaluating policy instruments is thus the two-and-a-half millennia old subject of political philosophy: what are just institutions for society or, which institutions are conducive to social justice.

Following Barry (1989, see his summaries, pp.359-62 and section 46) three main lines of reasoning for arriving at principles of social justice may be discerned. Number one sanctions a metaphysical or traditional acceptance of institutions as they are. Hindus base the principles of their institutions on religion, as less comprehensively, the state churches in some Western countries base theirs. For agnostics, the arguments given are not valid.

[1] The choice of environmental norms at the substance level would not be first choice since, in terms of the problem, it does not matter if the problem is caused by one substance or another. It is the empirically modelled total contribution to a problem that counts.

Institutions may also be cherished for being old and established, i.e. for reasons of tradition. As soon as the justness of existing institutions is questioned, tradition stops. In that case one or both of the two other main lines of reasoning is to be invoked.

Line number two bases principles for institutions on the mutual advantages of those under their rules. The advantages are measured against a state without these just institutions. A hypothetical social contract is assumed to move from the state of nature to the just institutions. If a Hobbesian state of war of all against all is assumed, any institutions which bring order would mean an improvement for those concerned and thus would be just. There is not much guidance then for the principles to be adopted. If the state of nature itself is already attractive, more specific principles are to be adopted. A second example of now modern contract theory is Gauthier, who sees competitive equilibrium as the state of nature. Cooperation in public institutions is only warranted if it can help solve the problems of external effects and collective goods (Barry, p.368). Utilitarianism, the moderate version of it, has a central place in this second line of reasoning. Institutions that are attractive to the members of society are just, especially if this holds for each member of society individually. Extreme utilitarianism is not included in this line of reasoning. It does not accept general principles for building institutions, as indicated in the preceding section, but requires the mutually advantageous choices to be made on a case level. From Plato and Aristoteles and the Sophists onwards these first two lines of reasoning to principles for society have co-existed.

Line number three, starting with the Stoics, bases institutions on the impartiality of the argumentation for them. In a sense it is a negation of number two in that institutions cannot be defended on the basis of the outcomes in the contest between personal interests. Kant's *Golden Rule* is a prime example of this impartiality. Rawls's *veil of ignorance* is another.

Only line number two and three seem relevant here for assessing changes in institutions - new instruments for environmental policy - as considered here. Barry (p.371) uses Rapoport's (1961) *Fights, games and debates* to further clarify the distinction between these latter two lines. Fights, as a non-institutionalized situation, are not relevant here. Games are situations in which persons with at least partially opposed interests interact, the rules of the game being ideally designed so that outcomes are advantageous to all parties. They belong to line number two, with each game representing one possible type of institution, for one type of situation. Debates are aimed at (mutually) convincing your opponents, using the knowledge one has of the situations oneselve and all others are in; that is line number three. Bargaining positions do not count in the process of rational argumentation. If institutionalised, they belong to line number two.

The stage has now been set for introducing more concrete principles, related to these latter two lines of reasoning in political philosophy. The three main value areas involved are first the utilitarian one, in line number two, requiring maximum individual utility and maximum social welfare based on individual utility. The first involved is a distribution-free principle on maximum available welfare, worked out for example in Paretian welfare theory. This is the domain of the *principle of maximum efficiency*.
In line number two a distributional aspect may also be involved, for example, if the utility derived from goods and services is assumed to be decreasing as they become increasingly

available to a person. A principle may then be formulated for institutionalising the redistribution of goods and services, e.g. income redistribution to the poorer sections of society, to effect a greater equality in that respect and hence a greater "total utility" or welfare. This is the first element of a group of *equality or distribution related principles*. In line number three, first, several other distribution related principles may be involved, not on the basis of their indirect contribution to welfare, but for reasons of a welfare-independent justice. The distribution related principles are all related to some form of equality or impartiality. A second element in line three is the freedom (or liberty)[1] that members of society can have. Various forms of freedom may be distinguished in this third group of *freedom related principles*.

These are the three main areas of principles distinguished here. They coincide with those of Meade (1993) as contained (in reversed order) in the title of his latest book: *Liberty, equality and efficiency*. Those that seem most relevant to a regulatory context will be specified. Before this is done, some further specification of the relevant context is due. Principles for institutions are discussed here not in a vacuum but in the context of Western democracy, with a mitigated version of capitalism. The main ideologies involved may have a broad spectrum though not as broad as it used to be. Especially after the collapse or transformation of virtually all communist economies, a broad consensus seems to be emerging on the desirability of a limited direct role of government in the economy. This consensus encompasses ultra liberals, such as Anderson and Leal (1991); the main stream of liberalism[2] since Adam Smith in the Eighteenth Century[3]; and an increasing portion of the modern socio-democratic movement, see for example Le Grand and Estrin (1989). Opposed to this strict delimitation of the task of governments vis-à-vis the economy there seem to be neo-corporatist ideas only. They are found primarily in Christian-democratic parties and, as a minority, in some conservative and socio-democratic parties. Their ordering of instruments is based on different principles, that are not investigated here[4].

A second aspect of context is that it is not only institutions that have to realise the principles. Within given institutions that may be conducive to the principles there still remains the realm of voluntary choice not based on direct self-interest, both individual and collective decisions on policy. These may play an indispensable role. Thus Meade (1973 p.52, as cited by Barry 1989, pp.394-5) states:

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[1] Freedom and liberty here are used as fully equivalent terms.

[2] Liberalism as an ideology should not be confused with ideas of the American 'liberals', mainly in the Democratic Party. They advocate a stronger government steering of the economy than do the Republicans.

[3] Highlights in this tradition are Bentham 1830 and Beard 1913, Commons 1924 and Hayek 1960. Marxist critique of liberalism denies the possibility of assessing societal efficiency. However, the mainstream of Marxism sees a capitalist phase as a necessary progressive stade in the societal development of productive forces, the latter implying some notion of efficiency.

[4] Neo-corporatism has been excluded from the analysis for several reasons, despite its attractive features as compared to the formalised litigious approach to (environmental) regulation. It can hardly lead to macro instruments, it is not in line with the principles that will be developed here and it would induce virtually uncontrollable complexity in the analysis by requiring feedback loops between government and society. Furthermore, in the realm of environmental policy the formal procedural approach, already dominant in the US, is steadily advancing in the European Community as well.

> *"In my view the ideal society would be one in which each citizen developed a real split personality, acting selfishly in the market place and altruistically in the ballot box.... It is, for example, only by such 'altruistic' political action that there can be any alleviation of 'poverty' in a society in which the poor are in a minority."*

Similarly, in environmental policy preserving environmental quality for future generations cannot be based only on institutions. For example, the definition of the contribution of a substance to global warming may be embodied in a "global warming tax". That global warming effect may last for centuries, thus taking into account effects on future generations. The level of the tax, based on a political decision, will decide the importance attached to such future effects. Thus the following two aspects of context are to be kept in mind:

◇ contributions are made to many principles in institutional contexts other than that of environmental policy instruments

◇ it is not only the right institutions that are required to realize the desirable principles.

The three main types of principles will now be treated in turn, leading to a description of six independent principles.

The principle of maximum efficiency can be formulated in a Paretian manner, as reaching the situation where nobody's welfare or utility can be improved without causing that of someone else to be worse. This principle of maximum welfare forms the basis for the "new welfare theory", now nearly three quarters of a century old, that did away with the previous foundations of welfare theory requiring ratio-scale defined utility and interpersonal comparisons of utility. If an institutional arrangement is to help approach Paretian optimality, several requirements should be fulfilled. Fully competitive markets, that imply some not very realistic assumptions such as absence of all scale advantages and full knowledge by all market parties of all technical aspects involved, would reach that maximum efficiency[1]. Even within the limitations that actually exist on markets, market prices would still give a reasonable indication of the relative costs of specific goods and services, as the value of other goods and services forgone by having those specific ones, that is for the lowest alternative costs. One central requirement for maximum efficiency then is that anything produced is to be produced for the lowest alternative costs possible. Pareto optimality and the more specific criterion of lowest costs possible are the first element on the list of principles for society in table 3.3.1, principle number one **(1*)**.

Principles of equality can be worked out in different ways. One way is related to utilitarianism, as one well-known form of justice-as-impartiality[2]. The utilitarian principle of equality of income is tempered by the negative effects on total income of excessive redistribution. Also, redistribution in the interests of equality is limited to maximizing total utility. It is this utilitarian principle of distribution that is listed in the

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[1] One main aspect of market failure forms the basis of this book: environmental external effects and the public-welfare character of the environment. Some advocate the quantified evaluation in welfare terms of environmental changes into project appraisal and general welfare assessments, thus incorporating environmental effects into the Pareto framework. See de Groot (1992b, chapter 10) for a survey of the arguments for this approach. It does not fit into the approach to policy development advocated here where environmental targets are specified separately, not hidden in a larger economic analysis.

[2] See Kymlicha 1991, chapter 2, on this utilitarianism, as one variant of Sen's *welfarism*. Welfarism, as a subjective utility oriented approach, also encompasses "Paretianism" of principle 1*.

table as principle number two (**2***). Rawls defines the principle as that redistribution of income that is still advantageous to those worst off. It is thus equivalent to principle one, but applied to a specific group.

Not only can the distribution of outcomes itself be the subject of principles, but also the procedures that can contribute to certain distributions of outcomes. If the latter aspect is dominant, it is only a derived principle that may be reduced to the combination of utilitarian principles two and one. However, the principle may also be based on impartiality in procedures. That formal-procedural aspect of equality, empirically related to efficiency, is the third principle in the table (**3***).

The following distributional principle is related to starting positions (Rawls 1972, p.7, not his constructed 'initial position' under the veil of ignorance). This may be interpreted again as a derived utilitarian principle, that would not add a new point of view, apart from the assumed empirical relation between the distribution of initial positions and the distribution of outcomes. There is also an interpretation based on impartiality, line three, where the right of one person, e.g. concerning a depletable substance or the use of a finite carrying capacity of the earth, is not less or greater than that of any other person. If this principle includes future persons, it covers inter-generational justice in initial positions as well. Distributional justice, its second variant, like equality in initial positions, is the fourth principle in the table (**4***).

Finally, there is an impartiality principle that stipulates fairness and equality in a substantial sense, as opposed to the purely procedural sense. It is the principle that concerns which cases should be considered equal; it is not principle 3* which deals with equal treatment. As a corollary, cases not equal in the substantive sense need not be treated equal. Race and belief are not allowed as such substantive principles. The motivation for killing another person is fully recognized as important in the institution of law for justly assessing such a killing. This substantive principle of justice, not worked out, is a fourth equality related principle, number five in table 3.3.1 (**5***).

Principles of freedom or liberty may be formulated in several ways. Gray (1991) has developed a taxonomy of conceptions of freedom[1]. He takes as a starting point the widely accepted definition of the concept of freedom by MacCallum. Freedom is a triadic concept: A person X is free from Y to do or be Z. This concept of freedom can be interpreted in many ways, expressing visions, values and areas of application. Gray distinguishes seven conceptions of freedom, four socially oriented and three personally oriented. The four social conceptions are freedom as:

◇ an absence from impediments
◇ availability of choices
◇ effective power
◇ status.

[1] With Gray as well there are only stilistic differences between the terms "freedom" and "liberty".

The absence of impediments is related to Rawls' (1970, p.60) Justice I, as the inverse, maximum freedom. Rawls defines the liberty principle as

> *"an equal right for each person to the most extensive basic liberty compatible with a similar basic liberty for others."*

The availability of choices has its parallel in Pareto optimality, as the maximum income allowing the broadest range of choices. That aspect of freedom, a line two type, has thus been covered already. Power and status are concepts of a relative nature. They have a zero sum character in that the increase of the power or status of one person simultaneously decreases that of one or more others. In that way they are related to distributional justice, see below.
Gray's three personal conceptions of freedom are freedom as self-determination, freedom as doing what one wants, and freedom as self mastery. These are not directly relevant in the context of regulation. They are not fully independent from the social freedoms. Institutional arrangements for the social aspects of freedom create the conditions under which personal freedom may flourish. This individual ethical part of freedom is not itself relevant for instrument evaluation.

The only remaining conception of freedom relevant for institutional design here is a social one, in Barry's line three. It corresponds to the principle of the greatest freedom that is compatible with a similar freedom for others, that is Justice I of Rawls. With Rawls, the freedom principle, his Justice I, cannot be traded for distributional principles. In the lexical order, its claims are to be satisfied first (p.244). Freedom as the absence of impediments is the one conception of freedom that has an independent relevance here, as the sixth principle in the table (**6***). It does not necessarily have prevalence over other principles here.

See table 3.3.1 for a survey of the six resulting societal principles. They seem to cover most positions taken in political philosophy. Some may not be relevant to all positions taken. For example, in the extreme liberal position, such as that taken by Nozick (see van der Veen 1991), all distributional related impediments to freedom are impermissible. Distributional principles of justice thus are not valid for Nozick. He not only sees any correction on outcomes of original endowments as unjust, contrary to Rawls. Also contrary to Rawls, he is against any correction of opportunities, that is against any correction of original positions in endowments or entitlements and against procedural corrections for more equal outcomes. The fully opposite position in this respect is to redistribute not only resulting incomes (as the outcomes of economic processes) for more equality, but also to correct for original endowments. See for example Pen and Tinbergen (1977) on compensating for acquired endowments and, as a thought experiment, on taxing innate endowments above a certain level. Any actual correction of outcomes, for reasons of justice, must be founded on one or more of the equality related principles.

3.3.3 Principles for environmental policy instruments

The six principles will now be treated in turn, through either the transformation of each one into a more operational form, applicable to the instrument discussion here, or its omission. An assessment is made on the role of environmental policy in all principles, and on the institutional arrangements available outside the domain of environmental policy specifically directed at realising these principles. The environmental principles are then

TABLE 3.3.1 PRINCIPLES FOR INSTITUTIONAL ARRANGEMENTS, AND FOR POLICIES, FOR SOCIETY AS A WHOLE.

Principles for society as a whole	
1* Pareto optimality:	minimal costs in making valued outputs
2* distributional justice 1:	equality in outcomes
3* procedural justice:	equal treatment, also related to equality in outcomes
4* distributional justice 2:	equality in starting positions
5* substantive justice:	being treated as equals, impartially, fairly and equally
6* maximum equal freedom:	lack of impediments

added, primarily based on the analysis of targets in Part Two. In the final part of this section the resulting double set of instrument-specific principles will be related to such other principles in the field of environmental policy, as sustainability and prevention, to see if anything relevant has been missed. The chapter ends with general conclusions about principles, in section 3.3.4.

Societal principles for instrument and policy evaluation

1 Pareto optimality*

The application of an instrument of environmental policy sets in motion a chain of events, on the one hand implying costs (including negative costs) and on the other, environmental improvements (including negative ones). Paretian welfare theory may assume that both costs and environmental effects, being valued effects, are included in the welfare analysis. It is assumed throughout this study, however, that environmental effects cannot generally be monetised and thus cannot be included operationally in welfare theory. The main reasons are the lack of a specified environmental effect chain; the collective nature of the goods and of the external effects concerned, and the incapacity of humans to compare different long-term developments in the environment, such as global warming in the year 2500, to current products, such as an extra piece of cheese in a sandwich. The welfare analysis thus cannot include the main welfare effects of environmental changes themselves. These effects somehow have to be expressed in their own terms. Efficiency then relates to two magnitudes, the environmental improvements, in their own terms, and the costs of arriving at them. Environmental efficiency or cost-effectiveness can be expressed as a ratio, as the amount of environmental improvement per unit of cost[1]. Costs are the alternative costs at the social level, in terms of prices that are based mainly on effectively established markets, but that exclude transfer payments.

In the context of environmental regulation the environmental improvements of an instrument are an "output" produced at certain costs, the social costs, as stated. To assess the efficiency of an instrument the numerator of the fraction should be specified as well.

[1] Efficiency is used in different ways in different contexts. Rawls, following T. Koopmans 1955, defines economic efficiency as total output for all inputs. Their maximum efficiency is fully equivalent to Paretian optimality. Koopman's efficiency here is 'maximum welfare'. I will use "efficiency" only in the sense of 'environmental/economic' efficiency. In a juridical context efficiency is often defined as effectiveness, comparable to the "efficient cause" in philosophy.

Environmental changes can be expressed in terms of the targets specified in Part Two. If an instrument has several independent targets, such as the emissions of several substances, the overall efficiency of its application cannot be established, except in the highly improbable case of the pure co-variation of all emissions. Furthermore, even if the target is limited, the efficiency measure should still somehow take into account all types of environmental interference not just that in the target. It is not possible to allocate the costs of some change in an economic process according to its contribution to the decrease of each substance. It is then impossible even to express the partial efficiencies for each substance. This problem is the same if the effects are expressed in terms of problems, but at a quantitatively different level. The number of problems related to an economic process will often be a tiny fraction of the number of substances related to it. Only when problems are ranked, through weights, will it generally become possible to assess the efficiency of instruments in terms of their contribution to "total environmental quality". This efficiency assessment will become possible even if the target of the instrument were still at a lower level of aggregation, through the weighted incorporation of all the environmental "side-effects".

How can the social costs of an instrument be assessed? A distinction is usually made between static efficiency and dynamic efficiency. The latter takes into account not only the adjustments that follow on the prime policy effect, but also broader effects on the functioning of the economic processes influenced, like the rigidity that is introduced by a policy instrument. If after the introduction of the instrument, dynamic effects occur, these will become dominant in the long run. If prohibitions diminish economic growth more than do financial instruments, they will be less efficient in the long run, even if these prohibiting instruments are initially more efficient. The latter, quite improbable, situation could occur if an omniscient and all-powerful regulator could introduce them, thus overcoming the tardiness of adjustments due to market influences on financial instruments. Dynamic efficiency cannot itself be measured. It can be predicted positively, on the basis of a central requirement for this type of efficiency. This requirement stipulates that in all choices related to a process or product the values sacrificed, that is the alternative costs, should be equal for the last unit of environmental improvement. Negatively, it can be measured as the impediments to making such choices.

Cost-effectiveness or *efficiency* is the first principle to receive a place on the list, though not necessarily with a top priority. It is first a characteristic of individual instruments. Combinations of different instruments, together making up a policy, might be evaluated on their aggregate efficiency as well.

2 Distributional justice in outcomes*

Instruments may differ in their effects on income distribution and could therefore be evaluated in these terms. There are, however, several reasons not to apply this principle to the evaluation of instruments of environmental policy. First, the income distribution in society is determined by many factors that can be influenced by public policies, especially by the increased education of the less educated, see Pen/Tinbergen (1977) and Pen (1990). Given a still unacceptable primal income distribution, redistribution is also a practical option. The tax structure and the structure of social security benefits have specifically been designed for this redistributional aim, as have many transfer payments from governments to households. Environmental instruments may differ in their

contribution to a more or a less equal distribution of income. Such differences are only of a quantitative nature, however, and can easily be corrected with specific instruments for income redistribution.
Secondly, effects occurring might be compensated. With the substance deposit and emission taxes, the additional public proceeds allow a reduction in other taxes or an increase in spending in other areas of government expenditure. The distributional effects of these adjustments might be designed in such a way that they compensate for the original adverse changes in income distribution, if any.
Thirdly, the evidence gathered on the distributive effects of environmental policies suggests that the net effect of current, primarily prohibiting instruments is near zero[1]. The insignificance of the effects would make the introduction of another principle a waste of intellectual capacity.

The equality of income distribution thus is not a relevant principle for instrument evaluation in environmental policy. The international distribution of the costs of environmental policy is a hotly debated issue[2], repeating the discussion on just institutions in a situation without a central authority. I prefer to avoid that debate here and rather assume the existence of a central authority, with all the powers of a state to implement environmental and other policies.

3 Procedural justice*
Justice regarding policies may be defined generally, as equal treatment of all individuals and organizations before the law on which the policies are based. Such an equal treatment is assumed for all the macro instruments considered, on the basis of safeguards in place outside the domain of environmental policy. This principle thus cannot play a role in the comparison of macro instruments here. It is relevant for assessing other types of instruments and the policies developed with them. Informal procedures as obtain in horizontal government would not be in line with this principle. An example is the case of milk packaging described in Part Five. General Electric Plastics is a major producer of polycarbonate and tried to market it for polycarbonate returnable milk bottles. That type of bottle probably has environmental advantages over both the polythene coated cartons and the glass bottles. GEP's political role in the Covenant on Packaging in the Netherlands was not strong enough to safeguard a reasonable procedure and outcome, so they retreated and lost their position. That type of instrument thus would be procedurally unjust.

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[1] The literature on the progressiveness or degressiveness of environmental policies is conflicting. The practical position taken here is that not much net influence is to be expected. See Merk 1988 for a survey and the analytics required. In a neo-classical model, adjustments would generally wipe out any prime changes in income distrubution induced by implementing new instruments. In an input-output model, with mainly fixed coefficients, such adjustments do not occur. The effects on income distrubution as modelled then are much larger, see for example Rose et al. 1987.

[2] A few examples may indicate the topics. Caldwell 1984 and Prittwitz 1984 define the subject and the organizational developments from an institutional point of view. Adede first states the necessity to integrate environmental policies into a broader international policy framework, especially that of development, and secondly points out that a more consistent and unified international legal structure should be formed, in which treaties will have a more systematic place. Burhenne 1993 describes international instruments, at the level of intentions and principles. Hoel 1991 explains how individual countries can influence total outcomes and their individual costs, from a rational actor gaming point of view. Ashworth and Papps develop a number of equity criteria from a purely normative point of view. Simonis 1992, finally, surveys the possible criteria for distributing the burden of reduction between countries.

4 Distributional justice in starting positions*

Justice in initial positions relates to the natural endowments persons receive independently of any efforts made by them. At any given time distribution should not exceed a certain maximum level of skewedness, to allow conditions for the just functioning of society to be met. Further limits on skewedness may be based on other aspects of equality. Especially in an international comparison inequality is now significant. There seems only one clear relation to policy instruments. Limiting for environmental reasons the use of resources devalues the initial endowments of those benefitting from their extraction now. However, the comparison of initial positions includes an intergenerational comparison as Rawls points out (p.287-8). The "veil of ignorance" extends to ignorance of one's generation as well. The intergenerational distribution that would be acceptable under the veil of ignorance is a just one. Not using a resource now leaves more resources as a future potential source of income or gratification. A greater inequality now, perhaps exceeding the limits of what is deemed just now, would be combined with a better temporal distribution, possibly helping to repair an otherwise unjust intergenerational initial distribution. It seems quite impossible to strike a balance in the relative justice of these two aspects of the initial position. Even if that were possible, it would still be difficult to connect that balance to specific policy instruments[1]. The distribution of environmental quality among generations is not determined by the instruments of policy but by its effectiveness. This analysis assumes the sets of instruments to be equally effective. Justice in initial positions is thus not a principle with which instruments of environmental policy can be evaluated here.

5 Substantive justice*

A government, with freedom of action similar to that of a person, should not only treat equal cases equally in formal procedures, according to principle 3*. It also might be obliged, on the basis of fairness, to treat equal cases equal in a substantive sense, as a principle that could even require adjustments in procedures. To put it the other way round, individuals (and groups) are entitled according to this principle to equal treatment by governments in a substantive sense. Substantial equality, in the context of environmental policy, is a partial mechanism because it does not involve total inequality but only a contribution to it. At an intuitive level it may first be described as the equal contribution to solving environmental problems. More precisely, substantive justice is defined here as the equal public treatment of equal (negative) contributions to environmental quality by the subjects involved. Requiring one person to reduce his emissions and another one not, or taxing his emissions and not doing so in the case of another person with similar emissions in a similar situation, is substantively unjust, regardless of effects on efficiency and income distribution. What "equal contributions to environmental quality" actually are, cannot generally be established now. As there is no overall principle or indicator of (differences in) environmental quality, equality can only be established partially.

[1] It might be possible to indicate in the life cycle analysis of products whether their effects occur in industrialized or non-industrialized countries. That could be the first step towards a broader analysis related to the distribution of initial endowments. There is olso a certain limited relationship between intergenerational justice in endowments and the way targets in instruments can be defined. The global warming contribution of some substance may be computed in a model based on twenty years or one hundred years of effects, both used by the IPCC, or for an indefinite period of time.

Examples are an equal contribution
◇ to an emission of some substance
◇ to the extraction of some resource
◇ to a single environmental problem.
If the relative importance of environmental problems could be established, the most general unit of "total environmental quality" could be used to assess equality. If the latter is not formally available, it might still be clear in many cases that treatment is manifestly unequal and thus unjust.

Equal treatment is preferably related to environmental equality in terms of a higher environmental target, at least problems instead of substances. Unequal treatment may relate to one instrument applied in different cases, or to a comparison of different instruments applied in one case. The latter comparison is not relevant. At the applied level of practical policies, the equality criterion applies only to the *sets of instruments* that make up that policy. If one instrument is only partially applicable, for example an emission tax covering only the monitorable emissions of a substance, the emissions not covered by it can be regulated by another instrument in the most similar manner, as reciprocal compensation, through for example an estimated emission tax. So, the equality aspect is first a characteristic of individual instruments applied at the same time to different cases. It may also be applied at the level of policies, allowing a correction of the inequalities of one instrument with the symmetrically inverted inequalities of another.

The nature of what constitute equal cases having been indicated, it remains to decide what should constitute the other element of substantive equality, equal treatment. The only options seem to be equal treatment in terms of the costs charged to regulatees and equal treatment in terms of the freedom taken away from regulatees. Both costs and freedom restrictions should be minimized, because of other principles (number 1 and 6 in table 3.3.2). The efficiency principle, however, has been defined in terms of social costs, regardless of who pays. For reasons of equality, the relevant type of costs would be the private costs of regulatees. Such a divergence between collective and individual levels does not exist with freedom. Therefore, equal treatment is defined here in terms of equal costs induced on regulatees. Equality then is the equality of the direct costs incurred by the regulatees in reducing their negative effects on the environment by a certain amount, including their private costs as transfer payments. As with efficiency, measuring equality is thus based on a ratio between costs and environmental improvements; it is an *equal trade-off* between regulatees. Measuring environmental improvements involves the same problems as measuring efficiency. In that case, however, the trade-off should be maximized, and it is also measured differently, especially by excluding transfer payments there..

How should the private costs induced by an instrument be determined? One might take into account all types of elasticity of supply and demand that together indicate the net effect resulting, in the short term or the long term, for all persons and organizations involved. The analysis then would be relative over the course of time, the assessment having a starting date and an ending date. Such an analysis would implicitly but unduly take into account the importance of products (through the elasticity of demand) and the financial resources of the regulatees and many environmentally irrelevant technical factors (through the elasticity of supply). What is relevant from an equality-of-instrument point of view is the direct costs that are inflicted on regulatees by one instrument, such as a "with-and-without" comparison and by a set of instruments, such as a comparison of these direct costs between instruments. The introduction of one instrument or another is the relevant situation to take into account. One further definitory choice is on the type of costs that are relevant, the total costs of applying the instrument integrally, or the marginal costs of one unit of environmental improvement, that is, the choice of taking all units of the denominator that change as a basis or only the last unit in the margin. It seems that from the normative point of view of instrument analysis, the normative impartiality of the instrument is at stake, abstracting from historical situations with regulatees as much as possible. It is therefore the marginal measure that is most relevant. Hence the equality of the application of one instrument or set of instruments is defined as *the equality of the marginal private costs of the last one unit of environmental improvement induced.*

It may be noted that the equality-as-impartiality principle has been defined here very restrictively. The definition does *not* take into account the financial resources of the regulatee, the usefulness of his products or his moral status. If such elements of principles entered the discussion on instruments it would be for other reasons than those relevant to environmental policy, or other principles might be involved that are not recognised as principles of justice here. Of course such aspects are very relevant for practical policy formation. Effects on poor but morally high-standing businesses with essential products, e.g. "farmers, the backbone of the nation," may constitute a legitimate reason for implementing *other* policies, such as the agricultural policies that have been institutionalised in most countries. Similarly, undesirable developments affecting income distribution may be corrected with the appropriate instruments for correcting income distribution, e.g. education and income taxes. Such arguments should not confuse the discussion on instruments for environmental policy.

It should also be noted that there now is a strong correspondence between efficiency and equality. Efficiency is a utilitarian criterion, derived from the maximum welfare principle. Equality is an impartiality criterion in its own moral right. The fact that in their operational dimensions they mainly differ in terms of transfer payments from and to government only is a happy coincidence, pointing them perhaps in the same direction. Since the impartiality criterion is measured in static (private type) costs, and the efficiency criterion in dynamic (social type) costs, their scores may sometimes vary considerably from one instrument to the other.

6 Maximum equal freedom*

The principle of maximum equal freedom can be interpreted quite plainly and straightforwardly in relation to instruments of environmental policy. First, the societal aspect implied by the compatibility with equal freedom for others is not relevant in the partial context of environmental regulation. The freedom realised cannot as a totality be specified in the analysis of policy instruments for environmental policy, let alone the compatibility of that total freedom with a similar freedom of others. What can be specified is the *contribution* of environmental policy *to* that *total freedom*. More precisely, the degree can be specified that these instruments subtract from it, as infringements on total liberty. Secondly, its content has been reduced here already to one aspect of freedom, the lack of impediments. The other main aspect, the availability of choices, is

TABLE 3.3.2 DERIVED SOCIETAL PRINCIPLES FOR THE EVALUATION OF INSTRUMENTS IN ENVIRONMENTAL POLICY

Principles for society as a whole		Derived principles for the evaluation of instruments in environmental policy
1* Pareto optimality: minimal costs in making valued outputs	→	maximum environmental cost-effectiveness = ***efficiency***
2* distributional justice 1: equality in outcomes	→	*not relevant* in choice of instruments for environmental policy
3* procedural justice: equal treatment, also related to equality in outcomes	→	formal equality, *assumed*
4* distributional justice 2: equality in starting positions	→	*not relevant* in choice of instruments for environmental policy
5* substantive justice: being treated as equals; impartially fairly and equally	→	justice as equal treatment of equal (changes in) environmental impacts = ***equality***
6* maximum equal freedom: lack of impediments	→	minimal impediments on freedom = ***freedom***

covered by the efficiency criterion. The freedom principle might cover the infringements on freedom through the environmental route, extending to future generations. How this should be assessed is a complicated problem. These environmental infringements are to be corrected by means that themselves limit freedom, through their administrative nature. Corrections, through instruments of environmental policy, should be compared to the lack of freedom they correct. However this balance is struck in particular institutional and cultural circumstances may be disregarded here. If an equal effect on the environment is assumed when comparing instruments according to the freedom principle, the assessment of long term environmental infringements on freedom becomes superfluous. Thus the measure for freedom is reduced to the impediments on liberty caused by the one instrument or group of instruments that effect a given environmental improvement. Technically, it is a ratio between infringements on freedom and environmental improvements, in the same way that efficiency was a ratio between costs and environmental improvements. It thus can be defined analogously as "environmental freedom-effectiveness". Since infringements on liberty cannot be measured at a ratio scale the quantification of that ratio does not make sense. Such infringements can be assessed on an ordinal scale at best. Thus, policies based more on instruments with fewer impediments for regulatees, environmental effects being equal, are preferred under this principle of freedom.

Environmental principles

Environmental principles have been grouped into three relatively independent value areas, the physical well-being of humans, or health; the quality of nature; and the functions of the environment in production and consumption. The question now is how these environmental principles relate to the foregoing general societal principles, if they are independent. Such a dependent relation is assumed by many economists to be embodied in the measure for welfare. That position will be rejected, although of course the negative value of some types of environmental problems may very well be expressed in monetary terms. An example is the corrosion of steel constructions caused by acid precipitation. The relation with equality could be more complex, since equality might also prescribe equal environmental quality between generations. Some options will be discussed. There does not seem to be a primary normative or empirical relation between the environmental principles and the societal principles of freedom[1] relevant in environmental policy instruments. If environmental principles remain independent from societal principles, the next question is how they can be made more operational. Being operational is not only required for defining instruments, but also for their assessment in relation to principles. Even if in that assessment environmental effects are omitted, under the assumption of equal effects on the environment, they are initially required in order to define what equal effects on the environment are.

Relations to welfare

Could environmental principles be reduced to a welfare principle? Two positions lead to such a reduction of environmental principles to a single principle, itself subsumed under the general welfare principle. First, there is the extreme individualism of neo-classical general equilibrium theory[2]. In welfare, only those things count that individuals value. Effects on all environmental value areas may be expressed in terms of a given numerary, e.g. the monetary equivalent of the value of some product. All values would have to be discounted to a certain base year. For the productive and consumptive functions of the environment this often seems possible[3] in principle. In practice, there are unsurmountable difficulties. A major problem is how to treat the long-term horizon of environmental effects. Depletion is the cumulative effect of hundreds or thousands of years of exploitation. An emission of some CFC may exert its direct influence on the ozone layer and climate for hundreds of years. In terms of physical well-being, especially health, even the theoretical connection is not clear. To approach the problem, people should have clear ideas on how to evaluate changes in their own health and that of others, including that of future generations. This is not the case generally. Introspection and observation of my direct social surroundings lead me to doubt that people know their mind in choices over pairs of situations involving different levels of health risks and different levels of income[4]. The quality of nature may be interpreted in terms of an independent moral entity, with rights to be respected. See de Groot (1992) for a survey. This precludes by definition any quantification in monetary terms. I would not go that far and prefer the point of view that the value is attached to the quality of nature by humans. That value of nature, alas, may not easily be reduced to a common denominator with values in other

[1] This is due to the restrictive definition of freedom. It is possible of course to define freedom as personal self-realization in terms of the personal relation to qualities of nature. I would prefer to exclude such types of individual related evaluations from the realm of public policies, to leave parts of the personal 'life world' free of authoritative collective definition. That of course is also a normative personal choice.

[2] Of the Paretian type, requiring ordinal preference ordering only.

[3] If damage caused by sulfur dioxide to crops and buildings is known and treated in financial terms, that part of environmental effects then should be disregarded in evaluating 'total environmental effects'. Damage prevented can be subtracted from the costs of emission reduction, giving a net economic total. This is the approach applied by Tinbergen in the first major cost-benefit analysis in the Netherlands, in the Fifties, on damming the major part of the combined estuaries in the Rhine and Meuse delta.

[4] A context can always be created in which people make some statement on their 'willingness to pay' or 'variating compensation'. The point is that the meaning of such statements is not clear, see the remarks on Sen and Scitovsky in the notes below.

domains, for the same reasons that make this impossible with the evaluation of health[1]. Such 'partial orderings' of states does not supply enough basis for the general equilibrium approach in economics. There is thus no basis for a single general monetary valuation[2].

The second position reducing environmental principles to a welfare principle is that a social welfare function of the Bergson (1937) type should be defined where each of the environmental effects distinguished receives a welfare score. A traditional item in such a Bergsonian social welfare function is the distribution of income. For such collective items, a collective decision may give the values. Under some assumptions quantification is possible and differences between states may again be expressed as a certain numerary. This is possible of course. It begs the question as to how, on what basis, the collective decision is made. It would at least require assessment in terms of the main value areas or related principles. If such an assessment is available, it could be incorporated into a general welfare function in a separate authoritative political (or non-authoritative personal) decision. Then efficiency, as dynamic cost-effectiveness, would reduce to maximum welfare. General emission taxes, their levels tuned to their contribution of each substance to the "total environmental problem", could realize this conversion. As a start, an operational definition of the total environmental problem would be required.

These arguments in favour of environmental value areas being treated as independent from welfare, equality and freedom ones may be supported by less theoretical, practical reasoning. If the environmental model specifying the full effect chains from all types of environmental interference to all relevant effects is lacking, the practical means for transforming scores in the three areas into a single welfare score is lacking as well. Environmental values related to instruments for environmental policy do not seem to be related to equality and freedom. At a general level there are intricate relations of course, the productive and consumptive meaning of the environment being directly related to welfare, as would be health, and equality being related to the intergenerational aspects of environmental values.

Environmental problems as principles

Thus the second question on environmental principles now is at what level they should be specified. In the discussion on targets for instruments, environmental problems were the

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1 This statement implies that there are domains where an individual ordering is not 'complete'; that in Sen's terminology, he has a quasi-ordering at best. See Sen 1969, pp. 8-9.

2 Mishan 1969 relates the "costs of economic growth" (his gloomy and influential complaint against the filth, stench, and belch as the side effects of modern mass consumption society) to missing markets for external effects. Amenity rights, if realizable through some instrument, could bend developments in more favourable directions. What is better is fully defined, albeit not in the same way by everyone. Internalization and separate facilities together may thus constitute a solution in Mishan, in line with most economic thinking since then. Translating environment into economic terms is mainly restricted by practicalities and theoretical sub-problems not yet solved, such as discounting the effects on future generations. Scitovsky's "joyless economy" on the other hand, while not denying the problem of external effects, takes a still gloomier view of human nature and human capacities, and the integrative possibilities for economics. The non-economic satisfactions of love, culture and well-being cannot easily be related to bread, TV-programmes and luxury holidays. The partial ranking of states makes rational decision-making extremely difficult. Following the beaten tracks, propagated in fashion and advertising, is an easy way out that unduly treats quasi-ranked items as ordered economic factors. With Mishan, "missing markets" may be constructed, giving "real" shadow prices as valuations in monetary terms. For a long time to come this will not be practically possible. With Scitovsky, market prices already extend their domain further than is sensible, thereby losing their meaning as a partial welfare indicator. Adding more partially ranked items would make the outcomes even less meaningful.

highest level of aggregation now deemed possible on the basis of operational empirical modelling. The three value areas then function as a qualitative background for the specification of these intermediary problem variables in the environment, such as global warming and acidification. At the other end of the effect chain, these problems also form the background for assessing the relative importance of contributions to different problems. A ranking of problems in terms of relative weights is practically required for choices on policies. If not made explicitly, it is at least implied. If made explicit in terms of weights, the sum total of effects on "environmental problems" could be translated in one normative step into effects on "total environmental quality". Thus a further intermediary step, at the level of the three different value areas, would be omitted. It seems the only present way to arrive practically at a general score on (total) environmental cost-effectiveness.

Which set of problems could be defined? This point will be treated more fully in Part Four, as part of the "classification" of the life cycle analysis of products. Here it should be sufficient to give the results obtained there. Environmental problems, as the operational principles for (non-local) environmental policy are:

Emission related:
- ◇ climate change
- ◇ ozone depletion
- ◇ human toxicity
- ◇ ecotoxicity
- ◇ acidification
- ◇ photo-oxidant formation
- ◇ over-nutrification

Extraction related, depletion of available:
- ◇ non-renewable resources, energy carriers
- ◇ non-renewable resources, minerals
- ◇ non-renewable resources, gene stocks
- ◇ renewable resources
- ◇ surface area

Other principles?

It may have been noted that principles familiar in the context of environmental policy are absent. These include sustainable development, prevention in general, waste prevention and recycling in particular, legitimacy, and "the polluter pays". Reasons for their exclusion here are given for each in turn. A few remarks on the polluter pays principle will be made in chapter 3.5.

Sustainable development as developed in "Our common future" relates to outcomes of policies. It covers both economic and environmental elements and would therefore appear a very good candidate to start with. It is not used as a general principle here for several reasons. Having one general principle suggests that subprinciples can be converted into it, especially that environmental and economic aspects might be merged. Such a merger is impossible and undesirable, as has been argued on several occasions. If such a formal relation of the main principle to its constituent parts is lacking, the use of the term may acquire a catch-all nature, obscuring more exact formulations of what in fact is concerned as principles. This is the position taken by Caldwell (1990) in his critique on Clark et al. (1986, his reference) who seem to have coined the phrase (p.177). He speaks of the "rhetorical trap of sustainable development", that constrains clear thinking on the subjects involved in the catch-all term. Since a ranking structure is suggested that is actually not existing, everyone may legitimately put in his own preferences. Some put more emphasis on the possibilities for continued economic growth, others on the limits that environmental variables put on that growth, and others on the quality of the environment that is to be preserved in the long run. Short-term problems, such as those of photo-oxidant formation, would be overlooked by the long term view analysis. The principles specified here together would cover the same subjects as covered by sustainable development. They even might be seen as building blocks towards making the umbrella term operational. The restriction of principles here to those that apply to instruments would make their range to small. Regularly occurring short-term problems would be included here, however, making their range too broad. The umbrella term sustainable development will therefore not be used here.

Prevention is a current declared policy principle, see the policy documents for the Netherlands (NMP) and for the US (State of the environment 1991, US Congress 1992). It relates to a shift from end-of-pipe control of emissions towards techniques and choices that indirectly prevent the substance from coming into being altogether or at least from becoming capable of being emitted. This characteristic of technical measures is not taken here as a principle. The actual occurrence of prevention is not excluded by macro-instruments, as in fact now is the case for many prohibitions directed at end-of-pipe measures. On the contrary, prevention may result, if advantageous, through the application of nearly all macro instruments. Put even stronger, there do not seem to exist other instruments for broad prevention policies than the macro ones specified here. Prevention will often be advantageous under the macro-instruments specified, especially since such preventive measures have so far not been extensively incorporated into environmental policies. There is nothing involved in preventative measures at a principle level that would make them better than end-of-pipe techniques. If end-of-pipe techniques are more efficient there is no reason for disregarding them. Thus there is no reason to take prevention as an independent principle for evaluating instruments.

Waste prevention and recycling also have been the explicit focus of policies. They are related to limiting both primary resource extraction with its emissions and final waste production. If taken as independent principles their use is quite dangerous. Waste prevention focuses on the input to one type of process, waste handling. Such a narrow focus may generally lead to the displacement of problems. Environmental aspects of waste handling are covered by the targets defined in instruments and by the principles formulated. Lifting out this single aspect does not seem to make much sense. If waste prevention is used more generally, as for example the prevention of environmental emissions into water, air and soil (a use apparently favoured by the US Congress in 1992, emissions = releases there), it is synonymous with emission reduction. It then only adds terminological complexity and might better be avoided.

Recycling, as a process induced by environmental policy, may or may not be attractive environmentally. The amounts of both primary production and the final waste of certain materials will indeed be limited by recycling it. If the supply of the material is very inelastic, as with cadmium, the primary production will not decrease much. The recycling processes themselves will also cause adverse environmental effects, through resource use and emissions. Economic value is a first measure of environmental effects, as long as a better analysis is not available. That quite a reasonable assumption indicates that recycling that costs more than virgin production may be expected to have greater adverse effects on

TABLE 3.3.3 PRINCIPLES FOR THE DESIGN AND EVALUATION OF INSTRUMENTS AND STRATEGIES FOR ENVIRONMENTAL POLICY

<table>
<tr><th>PRINCIPLE</th><th colspan="2">DESCRIPTION</th></tr>
<tr><td colspan="3">societal:</td></tr>
<tr><td>Efficiency</td><td colspan="2">Minimal costs of environmental improvements, dynamically expressed as improvements per unit of instrument? cost of the instrument(s)</td></tr>
<tr><td>Equality</td><td colspan="2">Equal treatment of equal assaults on environmental quality in terms of the marginal private costs imposed by the instrument(s)</td></tr>
<tr><td>Freedom</td><td colspan="2">The lack of impediments to behavioral choices by the policy instrument(s) applied</td></tr>
<tr><td colspan="3">environmental:</td></tr>
<tr><td>Health</td><td rowspan="3">Intermediary problems such as
*ozone layer depletion
*global warming
*acidification
*health risks
*over-nutrification
*resource depletion
*etc.</td><td>Minimizing threats to health and broader physical well-being</td></tr>
<tr><td>Quality of nature</td><td>Seeking highest quality of nature, as, e.g., in terms of naturalness and diversity at several geographical levels</td></tr>
<tr><td>Economic functions</td><td>Minimizing long-term impairment of functions of the environment in production and consumption</td></tr>
</table>

the environment as well. Only better analysis, e.g. life cycle analysis or substance flow analysis, may show whether recycling is attractive in specific cases or not. Waste prevention and recycling may or may not result when macro-instruments are applied. Instruments should not be evaluated according to their contribution to the techniques and processes involved, but to their overall attractiveness in terms of environmental improvement, cost-effectiveness, impartiality, and degree of freedom left to regulatees.

Legitimacy is often used as a primary criterion for assessing policies. It is assumed here that the factors creating legitimacy at the instrument level are closely related to freedom, efficiency and equality. At the combined level of policies a further factor is environmental effectiveness. Other factors contributing to legitimacy are legality and some general characteristics of government such as the charisma of its leaders. The legality of all instruments is assumed throughout. The charisma of officials is not a variable in the analysis of instruments. In the context specified here, legitimacy of environmental policy thus might be defined as the sum of the scores of the six principles defined. That definition could be very similar, perhaps fully coinciding, with a definition for sustainable development! It does not make much sense to use such an overall term for the same reasons as given when discussing sustainable development.

The conclusion on this mini-survey of other principles is that these are either covered, in a better substantiated way, by the principles that have been developed here, or do not add much more than terminological complexity, or simply are wrong as principles.

3.3.4 Conclusions

Principles are the main categories in terms of which instruments and the policies built on them can be evaluated. The fundamental changes implied in instrument choice for environmental policy require very general, high level principles. More specific principles may be derived by the addition of normative elements, by a derived evaluation with models, or by combinations of both. At the high level of abstraction required, models cannot generally be very formalized or quantified. A preference ordering of instruments cannot generally be based on an assessment of the specifics of its application. It is based on general principles and very general models. It is the "principle-guided choice" that may produce this preference ordering.

Six societal principles have been specified that might be relevant for assessing instruments and policies. Three of these still quite general principles have been selected and transformed for application at the instrument level: dynamic efficiency, substantive equality and freedom. The three distributional principles have been rejected as not relevant in evaluating environmental instruments and policies. Unwanted distributional effects, small if any, may be corrected by other, non-environmental instruments.
Three further principles specify what constitutes "environmental quality". They are human health, quality of nature per se, and the potential economic functions of the environment. Practically, these three environmental principles may be made operational in terms of a number of environmental problems that detract from them, such as ozone depletion and acidification. These problems may be transformed into a score for the "total environmental problem" by setting priorities in a weighted addition. Such priority weights have an individual ethical or collective normative meaning. Contrary to the common assumption among economists, environmental principles cannot be reduced to utilitarian social ones otherwise than by a normative decision. Here they are treated as being independent.

The resulting principles for evaluating instruments, three social and three environmental, are summarized in table 3.3.3 above.

Other principles are either covered by the principles specified here, such as prevention, or they are disregarded because they would not add much more here than terminological complexity, as is the case with sustainable development and legitimacy, or they are excluded because of being wrong, as is the case with waste prevention and recycling.

3.4 DESIGN STRATEGY: THE FLEXIBLE RESPONSE

3.4.1 Introduction

Principle-guided choice has been established in chapter two of Part Three as the most desirable and practical method for arriving at a reasoned choice of instruments. In the choice procedure only general aspects of government and society can play a role. Certain general assumptions are made. In government, a competent taxing organization, a competent but perhaps small environmental agency, and a judiciary able to cope with liability suits systemically are assumed. In society, a fully monetized economy is assumed, with markets influencing behaviour in nearly all processes of production, consumption, and waste handling. Markets need not be ideal markets but "reasonable" markets with some imperfections, but, e.g., with monopolies only as exceptions. There are of course externalities to public goods as further imperfections; they are primarily the environmental problems to be solved. Such general characteristics related to the implementation and functioning of macro instruments, and not only those, constitute the background here for the principle scores of instruments and instrument strategies for policies.

How can the principles be used to evaluate the macro instruments, is the first practical question here. A distinction is introduced between evaluating individual instruments and evaluating sets of instruments that together make up a policy. For such a policy, the choice between instruments does not affect environmental quality. If one instrument replaces another, all instruments may be adjusted quantitatively to yield the same environmental result. The specific contributions to environmental quality of a single instrument are not relevant, since the effects of all instruments together suffice. The preference ranking of individual instruments becomes independent from the absolute contribution to environmental quality of each instrument as applied in the set. Thus, the evaluation of individual instruments is fully independent of the environmental criteria[1]. These environmental criteria only apply at the level of the sets of instruments that make up a policy. They are required to quantify the instruments in the set and to establish whether two or more sets of instruments considered may attain the same environmental quality. Thus, also at the level of instrument sets, that is at the level of instrument strategies, the choice between these is not based on assessing them in terms of environmental principles. The societal principles selected can first be applied to the individual instruments, for evaluating them.

In environmental policies, several instruments may function together. The next question is how such sets of instruments can be formed, and what is their development strategy. First, the environmental criteria set certain requirements on their design. Then societal criteria may also apply again. Equality, for example, may be very restricted at the level of individual instruments. Higher levels can be reached at the level of policy, however, through complementary instrumentation. In that instrument strategy for policies the different lines of reasoning come together. The strategy is based on the model discussion,

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[1] The environmental criteria do enter the analysis in two other ways, in defining the targets of instruments and hidden in the societal criteria. Efficiency, e.g., is defined as "costs per unit of environmental improvement."

on the macro instruments developed, on the social principles applied to the instruments, and on the full set of principles applied to the combination of instruments in a policy.

The next section, 3.4.2, treats the preference ranking of the individual instruments for each societal principle separately. In the section 3.4.3 these are combined into a ranking of the instruments according to all societal principles together. The results are used in defining a strategy for policies in section 3.4.4. The chapter ends with conclusions in 3.4.5.

3.4.2 Preference ordering of macro policy instruments per principle

The preference ordering of instruments is based solely on their scores for the three societal criteria, efficiency, equality, and freedom. There is one complication in this procedure, however. For each main type of working mechanism, there are several instruments that differ in the type of object and target and, relatedly, in their level of aggregation. The ordering could be along two axes, within a given type the different instruments might be ordered, and the types could be ordered, or each individual instrument might be ordered. The latter approach is the relevant one but also the most complicated one. The instruments assessed until now encompass both the instruments gathered from the literature, especially the list from Bohm and Russell (table 2.4.6) and the added instruments, mainly macro ones, that have been discussed in the preceding section. See table 3.4.1 for a survey here. It includes a number of instruments that are not instruments for environmental policy in the strict sense. For very practical reasons, only the most macro instruments from the table, emphasised there, will be evaluated, see table 3.4.2. Thus, the problems of a two-step procedure, of first classifying all instruments per type and then between types, may be mainly avoided. The most macro instruments selected are all instruments for environmental policy in the strict sense. Other, less macro instruments will be "kept in reserve", if the more macro versions are not sufficient. It is interesting to note that current policy instruments are mainly those in the lower left block, in italics. One possible macro instrument will be omitted here. It is the improvement eco-audit. The first reason is practical again, that it has not been developed yet and will not be developed in this study. The second reason is that the specific application of the eco-audit as an improvement audit is covered by the only partial application of the life cycle analysis. That analysis runs from cradle-to-grave, while the eco-audit would run from cradle-to-product. There thus remain nine instruments to be ranked.

Efficiency

A quantified assessment of dynamic efficiency, defined as environmental improvement per unit of cost[1], requires a model of the societal functioning of the instruments concerned. Such specified models are lacking. The quantification of efficiency is not required, however, only an ordering of instruments according to their expected relative efficiency. This requires a qualitative assessment only. Efficiency is a characteristic of both instruments and policies. The latter is not independent however. It is a weighted average of the instrument efficiencies only. The constituting parts of the definition of efficiency,

[1] Improvements here are a decrease in negative effects on environmental quality.

TABLE 3.4.1 SURVEY OF INSTRUMENTS, CLASSIFIED ACCORDING TO MAIN TYPES (VERTICALLY) AND TO INCREASING LEVEL OF AGGREGATION (HORIZONTALLY)

Aggregation → *Instrument* *↓ types*	*1*	*2*	*3*	*4*
Structural instruments	*forced negotiations	*firm specific "extra strict" liability,	*location related sectoral extended liability	***extended liability** *(obligatory life cycle analysis, later possibly)
Cultural instruments	*moral suasion, esp. one substance - one product information	*all substances / all problems environmental information on products	*standard procedure for ecolabelling ***standard methodology for improvement eco-audit**	***standard methodology for problem oriented life cycle analysis**
Financial instruments 1: deposit-refund systems	*product deposit-refund system	*chemicals deposit-refund system per firm	*chemicals deposit-refund system per sector	***substance deposit on total emissions / one problem**
Financial instruments 2: taxes and pricing	*immission tax = firm differentiated emission tax on substance from one process/location *product tax (including input taxes)	*sector- uniform emission tax one substance / problem	***estimated emission / problem tax**	***uniform emission / problem tax**
Prohibiting instruments 1: tradable rights	*tradable immission permits per monitoring site	*general / regional tradable emission permits in excess	*general / regional tradable emission permits (as to reach immission standards everywhere)	***Tradable emission / problem permit, auctioned**
Prohibiting instruments 2: direct interventions	**technical operating/design standard per individual process = per firm* **technical design standards per process type / product*	*estimated one substance emission standard: per firm *idem: per process type = substance design standard	***estimated emissions standard / problem(s) standard per type of product or installation** *emission permit for all substances: for one individual firm	***estimated product standard, on life cycle based problem** ***general emissions standard / problems standard per type of product or installation**

TABLE 3.4.2 MACRO INSTRUMENTS TO BE ORDERED

Structural instruments	*	Extended liability
Cultural instruments	*	Standard methodology for problem oriented life cycle analysis of products or functions
	*	(Standard methodology for improvement eco-audit)
Financial instruments:		
1. deposit-refund systems	*	Substance deposit on total emission / one problem
2. taxing systems	*	Uniform emission tax / problem tax
	*	Estimated emission tax / problem tax
Prohibiting instruments:		
1: tradable rights	*	Tradable (auctioned) emission permit / problem permit
2. direct interventions	*	General emissions / problem standard per type of product or installation
	*	Estimated product standard, on life cycle based problems
	*	Estimated emissions / problems standard per type of product or installation

environmental improvements and costs, are the same as in equality. For equality, however, the trade-off between costs and environmental effects should be equal in all applications of one instrument and between several instruments. For the efficiency principle applied here, the environmental improvements per unit of cost should be as high as possible. At that highest possible level, the trade-offs should also be equal, for all applications of the one instrument being scored.

The efficiency of a certain policy-induced change may be modelled (quantitatively or qualitatively) in a limited period of time, with limited mechanisms included. This approach would result in only a partial picture of instrument efficiency, giving a comparative-static picture only. The dynamic efficiency relevant here is a concept that is much more difficult to assess. The difference between static and dynamic efficiency may be extreme, as the factors behind each may differ. The static analysis is restricted to the change envisaged in the instrument, usually specified in technical terms. The dynamic analysis primarily relates to effects on the speed and direction of technological change. A one time change will not be altered by a long-term horizon; its effects are not cumulative as are dynamic effects. Consider effects on technological progress as one main dynamic effect on productivity. Even a very small reduction of technological progress will become dominant as a cost factor in due course; its effects are cumulative. A tenth of a percent per year decrease in productivity growth becomes several percent of national income only after decades. Dynamic efficiency is superimposed on static efficiency. It may be expressed for a certain specified year, or as the integral over the whole period the time horizon covers.

How relevant is the potential dynamic effect of instruments? A clue could be found in the quantitative estimates of the slow-down in productivity growth due to environmental regulation. Several studies have estimated these dynamic costs. Christianson and Haveman (1983) surveyed the studies then available. A best guess for the dynamic effects of environmental regulation at that time was a drag of 0.3 percent on the annual growth of GNP. With direct environmental costs roughly at four percent, the dynamic effects are more important than the dynamic ones within a decade and a half. Jorgenson and Wilcoxen (1990) assess the slow-down in US economic growth at 0.19 percent per year for the 1974-1985 period. That is the same order of magnitude as found by Christianson and Haveman. Their thirty-five sector dynamic equilibrium model of the US economy incorporates substitution effects achieved through changing technologies and prices. There is no cultural element in the model. It has been fitted to data on the postwar period. They give no quantification of benefits. Hazilla and Kopp (1990) quantify the dynamic costs of environmental regulation, in a thirty-six sector dynamic equilibrium model, at a six percent decrease in growth cumulatively for 1970-1990, for both private consumption and GNP. Prohibiting policy measures have been introduced for each modelled year as actually implemented. As stringency of environmental regulations increases during that period, the yearly drag on economic growth will be over 0.3 percent now, according to their model. Their findings thus indicate the same order of magnitude for the drag on productivity growth by environmental regulation. As a general idea, the current type of regulation would slow down productivity growth by a few fractions of a percent a year. More stringent policies with the same instruments would drag down technological progress increasingly. The macro instruments developed here could subtract from these costly effects and might positively redirect technological progress in a more environmentally favourable direction.

That macro instruments could differ from direct interventions in terms of dynamic efficiency by a factor of five to ten, is a reasonable assumption. Their static efficiency would also be higher generally. The changeover from current instruments to more dynamic macro instruments could thus diminish the drag on efficiency growth by eighty to ninety percent, in the order of a few fractions of a percent per year. With current direct costs of environmental regulation at a few percent of national income, the material available thus suggests that the dynamic effects of a change of instruments would become more important than the direct effects within about a decade. For environmental policy, this is not a long-term horizon to take into account. Models of potential global warming usually have a time horizon of one hundred years. The conclusion here is that instruments should primarily be compared in terms of dynamic efficiency, with static efficiency playing only a secondary role.

For efficiency, there is a sharp division to be expected between direct interventions and all other main types of instruments. An extremely clever and powerful regulator (omniscient and omnipotent) might be able to design prohibitions with a static efficiency nearing, and sometimes even surpassing that of financial instruments. It is nearly impossible, however, to achieve a dynamic influence on economic actors resulting in a balanced environmentally oriented technological development. As indicated, it is more probable that the opposite will be the case as regulators are neither omniscient nor omnipotent. General technological growth will be impeded and environmentally oriented technologies are not stimulated. This places the three direct intervention type prohibiting instruments at the bottom. They do not differ much in efficiency. The order according to efficiency of these three instruments is the general emission design standard at the top, being based on real measurements, the LCA based product standard in the middle, and the estimated emissions design standard the bottom, at positions 7, 8 and 9 respectively.

Cultural instruments, if interpreted as having no costs, are at the top; their environmental improvements are for free. People include environmental considerations in their personal preference functions and choose what they prefer most. These preference functions form the basis for the economic analysis of efficiency. If not totally for free, at least the improvements will be highly efficient, more so than those of financial instruments. The same factor that makes them so efficient also restricts their importance. Cultural instruments only cause clear environmental improvements with low costs per unit of improvement. They work only at the top of the pyramid (or Eiffel tower) of improvement options, ranked according to efficiency. So the standard methodology for LCA is number one in efficiency.

With financial instruments, the choice on the trade-off is an explicit public decision, related to the amount of emission reduction desired. The trade-off, in the margin, is fixed in the level of the substance deposit and the emission tax. Economic decisions with a higher trade-off than that level all are profitable, both in production and consumption. Their trade-off will be set lower than that of cultural instruments, and, therefore, they lead to broader effects. The broader applicability places the substance deposit slightly higher than the emission tax, in second and third place respectively. The near-ideal estimated emission tax comes next, assuming that the estimates come close to the real emissions.

With the tradable emission permit, the trade-off results from market forces. It will not differ much from that of the emission tax and the substance deposit, if roughly the same emission reduction is to result and the total of all permits is set at that level. The long-term predictability of the permit price is lower than with emission taxes. The political risks of change are similar, but there is an extra technological risk. This uncertainty somewhat limits the possibilities for rational technology design, as compared with emission tax and substance deposit. It has been placed slightly under the estimated emission tax, at place five.

Extended liability is most difficult to assess in terms of the trade-off induced. On the one hand, the dynamic effects of preventing the expected level of liability costs are undisputed, in the same order as those of financial instruments. Several effects subtract from this efficiency. First, the damage only takes into account a tiny fraction of all environmental effects caused, those to which causes and costs can be assigned. Secondly, the damage amount as assessed in "similar" cases is highly variable and difficult to interpret. It relates to site-specific effects, while all other instruments have generally defined targets, abstracting from these site-specific effects. This complicates the application of the results of cases to a given situation, diminishing the preventive, dynamic effects of the instrument, and perhaps also leading to overreaction in other cases. Furthermore, the highly unpredictable nature of the outcome of the judiciary process may lead to severe under reaction or over reaction. See in this sense Bardach and Kagan (1982, the chapter on litigation) and especially Kagan (1991). Thirdly, unlike financial instruments, there is a delayed payment, lowering the discounted value of the costs[1]. Fourthly, if there is a tendency for long-term risk taking, as may be expected, the already considerable discounting of future costs will increase further. Fifthly, there is the cultural mechanism mentioned already that extended liability, excluding all negligence, removes the normative content from liability. Being liable no longer means that one has done wrong in a moral sense. The broad preventive working of such a normative attitude is hard to assess but could be extremely important. Finally, there is a more complex argument based on the structural effects in society which run counter to flexibility and dynamic change. The cost in man-power of traditional liability, based on fault, has never been substantial. However, the cost of litigation in extended liability may be very high. In the US the costs of litigation in soil contamination cases have been extremely high, about as high as the assessed damages. These high costs might be seen as temporary, due to the provisional status of these new forms of liability. They may also be a step towards an excessively juridical society, see Donner (1988), with further high costs hidden in the private procedures to forego litigation[2]. Any change in technology then becomes a risk that should be assessed meticulously beforehand, including those that are environmentally beneficial. These arguments together place extended liability somewhere between tradable rights and direct interventions, at sixth place.

Another type of reasoning abstracts from differences in the societal functioning of instrument but assesses the potential efficiency of instruments on basis of only their

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[1] Assuming equal targets and price levels per unit of environmental damage a major difference between liability and taxes would be the time of payment. With taxes and deposits, payment takes place at the start of the environmental processes set in motion, with liability at the end. That difference may be several years. The effects on climate of a CFC emission now lasts for centuries. Let us assume an average delay in effects of thirty-five years. With damage amounts set equal between instruments, and real interest rates at around four percent, the discounted costs of litigation would be reduced to about twenty five percent of those of direct financial instruments.

[2] Even Western societies differ strongly in their levels of litigation. Taking the number of lawyers as an indication, there are substantial differences between such closely related countries as Belgium, the Netherlands and Germany, and between these and California, for example.

target. Instruments whose target is nearer the final values affected waste fewer costs in inevitable side effects, assuming a similar environmental effectiveness. Extended liability instruments then are prime choice, the target being an assessment in monetary terms of all values affected. For most supra-local problems, however, the instrument is simply not applicable. Everybody emits CO_2 and everybody, including a number of future generations, is affected by the global warming effects of these emissions. Practically, the efficiency is thus severely reduced if compared to a target such as "total environmental effects". Life cycle analysis, especially if it weighs problems into a single score on "total environment", therefore take first place. Substance deposits relating all substances to one problem are in second position. However, if a system of substance deposits were developed for all substances relating to all environmental problems, that type of deposit would rank equal to the system of life cycle analysis. The same holds for integrated systems of emission taxes. Estimated emission taxes come close, as long as the estimate comes close to real emissions, and may similarly be transformed into estimated problem taxes.

Such integrated, problem related systems seem hardly realizable with tradable permits. Design standards, fixing certain technology, that is, all three types of prohibitions included on the list, are clearly last in this exclusively target oriented efficiency analysis. Variations in aggregation within types of instruments do not relate to the efficiency based on targets, as long as their targets remain the same. This is possible only to a limited extent. The only target related efficiency analysis does not show a marked difference from the more general analysis.

Combined, the two types of reasoning result in an efficiency ranking with cultural instruments on top, financial instruments and tradable rights following, then the structural instrument of extended liability, very provisionally, and with direct interventions clearly in bottom position. The broader the application of instruments is, for a given target and a given instrument type, the stronger will be their dynamic effects. It thus seems reasonable to expect increasing efficiency as the level of aggregation increases. The following ranking would result for *efficiency*:

* Extended liability	**(6)**
* Methodology for LCA	**1**
* Substance deposit	**2**
* Uniform emission tax	**3**
* Estimated emission tax	**4**
* Uniform tradable emission permit	**5**
* General emission design standard	**7**
* Estimated product standard, LCA	**8**
* Estimated emission design standard	**9**

Equality

Equality, defined as equal treatment of equal (differences in) detrimental effects on environmental quality, is a criterion which, like efficiency, combines an environmental variable and an economic variable. Equal treatment implies an *equal trade-off* [1] between

[1] Equality and efficiency may seem to approach each other closely this way, but the overlap is restricted to static efficiency. Dynamic efficiency, a much more important factor in the long run, is largely independent from equality. Equality may be fully realized with a prohibiting instrument such as technical standards, assuming an omniscient

costs incurred because of an instrument or policy and the contribution to environmental quality realized. The different types of instruments differ quite fundamentally in who chooses this trade-off and how high it will be. Equality is defined only if the trade-off is defined. Therefore a measure of costs and of contributions to environmental quality are required. Costs are direct alternative costs in market terms, of the static type. On the environmental side things are more complicated. A comparison of the environmental effects among the different applications of a given instrument, and among different instruments, would ideally show effects on "total environmental quality", such as the weighted total of the contributions to all environmental problems. If such a measure does not exist, the full trade-off is not defined, nor is equality as already indicated similarly for efficiency. If partial measures on environmental quality are used, such as a certain individual environmental problem or emission, comparisons on equality are at best partial. Problem shifting to other substances and other problems can then readily occur.

With extended liability, as the *structural instrument*, the level of the trade-off could be chosen by judges. This might be only implicit, since the costs of damage prevention should not play a role in assessing the damages. However, if a glimmer of negligence still plays a role in assessing damage levels, judges may use the level of charges incurred in preventing the damage as an argument in their quantification. Generally, however, the decisions of judges will be restricted to quantifying the environmental part of the equation. The second part of the trade-off will be decided by private firms. The expected costs of liability (= the level set by a judge plus other private costs times the chance that payment will be made[1]) per unit of damage will set the level of the costs incurred in damage prevention. Since judges use only a very partial measure of environmental effects, their decisions inevitably disregard the effects on other environmental variables. Also, there will be differences between judges in the level of damages set in comparable cases. Both factors limit the maximum equality that can be realized.

Less aggregate variants of structural instruments score less on equality. Sectoral liability rules, as on waste processing, or oil drilling, will lead to different results in terms of damage quantification. Firm-specific damage quantification functions built into the permit on their operations (see the example of US offshore oil exploitation), will diminish inequality between judges. It will increase that between areas of application, each with its own damage function[2]. In all structural instruments the level of the trade-off is not decided, explicitly or implicitly, by government. It is a combined decision by judges and by private persons and organizations. General extended liability could in court practice be broken down into specialized sub-fields, the assessment of damage amounts differing between areas of application.

For the standard method for life cycle analysis, the most aggregate *cultural instrument*, there is again no explicit trade-off. There is even the point of view that a trade-off cannot be defined since the improvements do not cause costs. Consumers develop their

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regulator. Its dynamic efficiency, however, will be minimal.

1 Corrected by a factor for risk aversion.

2 The general specification of one damage function in all extended liability, including purely collective damages, could create equal treatment. That function makes it possible to collect damage payment before all damages have actually occurred. Governments would be the main recipients for these damages. The problem then is how to define the frequency of the court case, once a month, once a year? If, as a mental exercise a case were made per unit emitted, something very similar to a "total problem" tax would result. The only difference would be that judges, instead of governments set the level of the taxes.

preferences in consideration of any factor they want, as they like. This consumer sovereignty is the basis for all welfare computation. Thus, if they buy the environmentally better option they are following their own preferences and there are no costs. If, however, consumers are seen as producers of utility, having their own utility production function, they can compute costs against advantages and a trade-off may then be defined. Let us assume here that private decisions do involve a ranking of costs and expected environmental improvements, an arguable position. Only persons and organizations make the trade-off between costs and effects. They may adjust their behaviour assuming that all others will act similarly. Or they may assume that they would be playing the fool alone, surrounded by free riders. In the latter case there would hardly be any behavioral change[1]. No costs are incurred and no improvement is realized. Reality will be somewhere in between, with private decisions understating the individual trade-off made in principle. Both the level of the ideal personal trade-off and the degree of understatement in practical choices will differ. The equality of the instrument will not then be maximal. The comparison with structural instruments does not give a clear result. Since only the highest trade-offs will be effective, the differences in trade-off cannot become very great. If the LCA is in terms of "total problem contribution", it probably would be superior because of the very partial measure on environmental effects in liability. The "separate problems" oriented type would not generally give clear comparative results. Grosso modo, cultural instruments would score better on equality than extended liability.

The substance deposit is the most aggregate *financial instrument*. As with all financial instruments, the trade-off is decided collectively and is more or less equal at the margin. It is the only instrument type where the trade-off is unequivocally and publicly defined, in cost per unit of emission or per unit of environmental problem. The equality in terms of "total problem" is safeguarded if, first, levels of problem deposit/taxes were set in relation to their contribution to the total problem, and the derived level of substance deposits/taxes would reflect the contributions of each substance to all problems.

Uniform emission taxes have only a slightly lower level of equality than the substance deposit, the only difference being their less broad practical application. The estimated emission tax scores a bit lower again, because some inequality will result from the "noise" generated by the estimating procedure. These financial instruments score much better on equality than the cultural and structural instrument.

With the uniform tradable emission permit, as the *first type of prohibiting instrument*, there is no explicit trade-off. Implicitly, the choice of the amount of permissible emissions fixes the trade-off. Through the market created, a price per unit of emission results for the trade-off, given a certain level of demand for the emission right. Thus an explicit trade-off ratio does result, but only indirectly. As demand for emission rights changes so does their price level and so does the trade-off. This causes inequality in the course of time. It is also improbable that tradable substance emission rights can fluctuate in the same way for all substances. Fluctuations would thus create inequalities in the

[1] Limited empirical evidence indicates that a substantial proportion of the consumer population will adjust its behaviour environmentally if costs are low. Circumstantial evidence shows that designers are willing to incorporate environmental considerations into their design. Most design schools have introduced courses on environmental design. The famous French designer Philippe Starck explicitly stated that his designs take the environment into account by using minimal amounts of only common materials.

contribution to one problem, to different problems and hence to the total problem as well. Defining "tradable problem rights" would partly solve this problem, i.e., only the first step. Tradable "total problem contribution rights", relating all substances to each other, would solve it fully. Fluctuations in time also would be flattened out by such a highly aggregate unit of trade. That option would seem out of reach for a long time to come, however. The score on equality will not be much lower than that of the financial instruments and higher than both the cultural and structural instrument.

With the three types of standards, as the *second type of prohibiting instrument*, no explicit trade-off is given directly, and no explicit trade-off results indirectly. General design standards, for the emissions from one product or installation, are specified by different authorities. Costs incurred by firms for following the standards will differ and can hardly be known to the regulative bodies. As technologies develop, the costs will change. The implicit trade-off, which is always present, will thus differ highly between the same technology used in firms and between different technologies. It will also fluctuate substantially in time. Less aggregate instruments will lead to an even greater inequality.

With standards, the administration may or may not have specific trade-off ratios in mind when allowable emissions, actually measured and estimated for each installation, or for each life cycle, are set. If so, this could improve equality somewhat. Lack of information and dynamic autonomous development in technologies would still result in considerable trade-off inequalities. Even then, the standards score lowest on equality.

A first point to note is the very different ways the trade-off is determined and functions with the various types of instruments. With structural instruments it is judges and private persons and organizations that together create a partly public, reasonably uniform trade-off. With cultural instruments the trade-off is a purely implicit, private affair, differing between individuals, and probably also within individuals for different choices. With direct interventions the trade-off is also a purely implicit, but now public affair, differing between authorities, and also between the decisions of a given authority. With tradable rights, an explicit trade-off results in the permit market, changing in time and differing between substances and problems. Only the financial instruments have an explicit, publicly decided trade-off, that may be stable over the course of time. Its substance levels may be attuned to problems. If this procedure has been followed for all problems, problems will in fact have been ranked and taxes would then be "total problem taxes".

The ranking order of the macro instruments as to *equality* is as follows:

* Extended liability	**6**
* Methodology for LCA	**5**
* Substance deposit	**1**
* Uniform emission tax	**2**
* Estimated emission tax	**3**
* Uniform tradable emission permit	**4**
* General emission design standard	**7**
* Estimated product standard, LCA	**8**
* Estimated emission design standard	**9**

Some types of inequality at the instrument level may be compensated for at the level of policies, that is as sets of instruments. For example, an emission tax could be levied on the emissions of some substance not covered by a substance deposit. In that case a certain measure of complementarity in potential application is accomplished, leading to a better equality for the sum of the two instruments.

Freedom
Freedom, as the absence of impediments by governments on behavioral choices, can be assessed in terms of the behavioral choices made impossible, or added. Not much modelling on how the instruments work in society is involved. This principle is broadly applicable for comparing instruments, both within a given type and among main types. It is not applicable to policies in the form of instrument sets, in a way other than as the sum of the freedom taken by each instrument individually. Lack of freedom in one instrument cannot be offset by other policy instruments that leave much freedom, as was possible with equality.

There is a clear division between the prohibiting standards and all other main instrument types. Even less aggregate tradable rights and emission taxes could hardly result in the same amount of freedom taken as that of enforced behavioral specification of standards. That comparison is quite straightforward.
The structural rules regarding property are there to safeguard individual liberties against infringements. In this broader sense of freedom, they even create freedom. By extending liability, governments do not in any way limit freedom as defined here and might even extend it. Cultural instruments, here the public availability of a methodology of environmental life cycle analysis of products, also takes away little if any freedom. By showing hitherto unknown effects of choices one might even argue that freedom (in a slightly broader sense than used here) increases. Financial instruments such as the substance deposit perhaps limit freedom more since boundaries must be drawn specifying payment obligations and refund rights. Similarly, emission taxes and tradable permits require technical facilities for the measurement and control of emissions. Tradable permits additionally fix the administration of the trade allowed, with technical boundaries creating some rigidity.

The resulting ranking is quite in line with the Zijlstra classification of instruments. The Zijlstra typology itself was ordered according to the degree of freedom taken from persons and organizations. Instruments can therefore be ranked according to *freedom* as follows:

* Extended liability	**1**
* Methodology for LCA	**2**
* Substance deposit	**3**
* Uniform emission tax	**4**
* Estimated emission tax	**5**
* Uniform tradable emission permit	**6**
* General emission design standard	**7**
* Estimated product standard, LCA	**8**
* Estimated emission design standard	**9**

3.4.3 Preference ranking of policy instruments: combined results
The combined assessment of the instruments on all three principles shows a high degree of co-variation. Only the scores on extended liability and the methodology for LCA differ for different principles, see the survey in table 3.4.3 below, with the deviating scores underlined. For the remaining instruments there is a simple dominance relation, with all scores dropping from substance deposit to estimated emission design standard. The

remaining question on ranking relate to the main score differences, the low score for the LCA methodology in terms of equality, as compared to its other scores, and the high position of extended liability on freedom.

The problem associated with extended liability is that the efficiency for the effects it covers may be high, as high as that of financial instruments. But it can cover only a fraction of all environmental effects and it covers these in a mainly site-specific way, contrary to all other instruments. There are also many afterthoughts on its efficiency, as indicated especially by Kagan. However, in this respect it might still be better than prohibitions. It also scores better on equality than these latter instruments. On freedom it scores better than all other instruments. Together, the arguments against its use and the contradicting scores resulting indicate such a special position that no total ranking is given.

The methodology for life cycle analysis scores first on freedom and efficiency, and relatively low on equality. Following Rawls, and certainly such liberals as Nozick and Anderson, precedence might be given to the freedom aspect. Even not going that far, there are still reasons to give it a high place in the overall classification. The scores of the instruments on equality fall into two widely diverging but internally homogeneous groups, the lower group of direct interventions and the higher group of all other macro instruments. Only within this latter group does the LCA methodology score low. Its relative score on efficiency is extremely high, also in comparison to financial instruments. Compared to financial instruments its score on freedom is also very high. If the three principles are ranked equal in importance with equality and are no more important than the other two, this indication of quantitative results would lead to first position in the overall score. Either following Rawls and Nozick, or following the suggested ranked scores for principles, the LCA methodology takes number one place.

Hence the following total ranking results. Extended liability is somewhere in the higher regions but without a specific place. There are doubts on the attractiveness of its functioning. Long-term historical developments, however, indicate its increasing prevalence. Because of its site- specific application to cases, it might have an unexpected role in preventing excessively high local concentrations, together with other locally differentiated instruments.
The methodology for life cycle analysis is a first choice or at least in any reasonable evaluation, very near the top. Especially if normative weights on problems are added, it may have a very broad influence, if limited to the choice of situation.
The substance deposit and the equivalent but less broadly applicable emission tax are second and third. Each could be set up as a problem oriented system, applying to each substance, the deposit or tax level related to its contribution to a given problem, or, preferably to several or all problems. Such a system of related taxes would quite explicitly define the ranking of problems, combined with an assessment of the problem contribution of each substance.
The estimated emission tax clearly is a second choice financial instrument, taking fourth place. Emission do not even constitute the complete target, which is a specific technology or procedure. However, the differences between first and second choice are fluid and may be minimized. Emissions of a substance in a larger flow may be measured discontinuously as concentrations, with additional data provided on total flows. A tax based on such data would still be an emission tax. A periodic product tax, e.g. on the use of a car could be based on actual emissions per kilometre, measured once per period in a standard test run,

multiplied by the number of kilometres driven during that period. This would be an estimated emission tax if the period is a year. It would be an emission tax if measurement took place every minute. Somewhere in between is the change from a second choice to a first choice financial instrument. This second choice instrument has to be rated against the most aggregated types of prohibitions such as tradable rights, the uniform tradable emission permit. A conditional choice places the estimated emission tax fourth, assuming only those applications that closely approach the full emission tax.
The nationally, or even supranationally, tradable emission permit is fifth. It is not the US type, but the freely tradable one, with actual emission measurement for all installations requiring permits. To bring the tradable permit in line with the financial instruments, it could be set up as an auctioned permit, with free trading after purchase. The primary right as thus auctioned would be one of collective property. The form and frequency of the auction and the period of validity may all be chosen so that the resulting market is relatively free. One way to enlarge the size of the market would be for governments to sell "problem contribution permits" instead of substance emission permits. Instead of a right to emit NO_x and a right to emit SO_2, the right to emit "acid equivalents" could thus be traded. Similarly, the amounts of allowable heavy metal emissions could be expressed in terms of toxicity equivalents, and not per individual metal.
General design standards may be differentiated between those requiring actual emission measurement, the general design emission standard, and those where emissions are estimated indirectly, the estimated emission standard and the estimated LCA score. The general design emission standard is sixth. It specifies how much certain types of installations may emit, differentiated to the size of their production. It differs from the estimated emission standard, in eighth place, in that the latter does not require actual measurement. Its environmental performance is supposedly indicated by its nature, as in the case of allowing cars on the road that conform to exhaust standards in a test run. In between these two instruments is the estimated product standard, LCA-type, with its target aggregated at least to the level of problems.
See the survey on the preference ranking of instruments in table 3.4.3 below.

TABLE 3.4.3 PREFERENCE RANKING OF POLICY INSTRUMENTS ACCORDING TO EFFICIENCY, EQUALITY, AND FREEDOM, AND TOTAL PREFERENCE RANKING

	efficiency	*equality*	*freedom*	***total***
* Extended liability	(6)	6	1	**(?)**
* Methodology for LCA	1	5	2	**1**
* Substance deposit	2	1	3	**2**
* Uniform emission tax	3	2	4	**3**
* Estimated emission tax	4	3	5	**4**
* Uniform tradable emission permit	5	4	6	**5**
* General emissions standard	7	7	7	**6**
* Estimated product standard, LCA	8	8	8	**7**
* Estimated emissions standard	9	9	9	**8**

A overly restricted list?
The question now is whether, apart from local, site-specific problems that are certain to require separate instruments, these macro-instruments together are all instruments required for establishing a generally sufficient environmental quality. If not, other instruments would have to be added. Which instruments would be on the candidate list? The higher level types, the less aggregate "second choice" variants, would all have to be considered.

Liability rules for specific effects, such as those of solid waste or chemical waste, might be considered. These instruments have been introduced in several Western countries, with noticeable effects in the US and the Netherlands. Their structure complicates general liability. Their restricted domain of application reduces the general disadvantages of extended liability. Given the already ambiguous nature of extended liability some specific, less aggregate structural variants might be included, in the spirit of the American extra-strict liability defined in permits. No such systems will be investigated here.
Also, systems of ecolabelling might be added as a cultural instrument. If these systems are fully based on the standard LCA methods, there hardly is any reason to have such an additional programme. Consumer organizations and environmental organizations are quite able to produce and disseminate the comparative studies. Producers are quite able to use these results in their advertising and more general public relations. For this reason, a system of ecolabelling is not to be included in the group of instruments for macro-environmental policy design.
The theory on the improvement eco-audit, if further specified, might establish its place as an independent instrument. That further specification will not be done here.
Of the second choice financial instruments, only product taxes have a general application. The most attractive versions of product taxes have already been implied in the substance deposit (e.g. a carbon tax on fuels combined with repayment upon CO_2 storage, (see the case on energy depletion and global warming) and the estimated emission tax (the car example above). The remaining product taxes, including input taxes, are hardly related to specific emissions. They would find a place after the individual emission permits currently being developed in several countries. Further financial instruments directly descend to the level of the individual firm and are not attractive. Non-regulative additional elements to financial instruments, such as compensation programmes for those damaged, are not the concern here.
Less aggregate direct prohibitions are even less promising for macro policy design and would thus be considered only if all other more macro instruments, together, failed to achieve the desired environmental improvements.

Subject to the minor qualifications given, the eight instruments chosen seem to form a complete set. Other instruments do not add much more than complexity. There thus seems no compelling reason now to enlarge the list of instruments for macro-environmental policy design.

3.4.4 Design procedure: the flexible response strategy

How could the design of macro-environmental policy be approached practically? Which instruments are to be applied where? There is one main point on which the structural and cultural instruments differ from all others. Contrary to all other instruments, the choice to implement the methodology for LCA or extended liability is a dichotomous one. Either they are introduced for all cases, or for none. In this sense they are "collective" instruments. There is no choice to be made on their application in specific domains or cases. They apply or do not apply to such situations but require no extra policy decisions (apart from court decisions, which here are not seen as administrative implementation).
These two can be established regardless any time. Just go ahead and institute them, preferably on a broad international scale. Their functioning in specific situations and their general influence on behaviour do not depend on differentiations in policies. Quite the contrary, their general preventive effects depend on their uniformity. The methodology for life cycle analysis is a must for environmental policy. At virtually no cost broad,

highly efficient environmental improvements will result[1]. As indicated, extended liability has been half-heartedly incorporated into the design. The attractiveness of this instrument is far from certain. It has been included more because it is simply becoming established than because it is a top choice. The introduction of the macro-structural and the macro-cultural instrument are thus both case-independent. Any policy strategy would always start with these.

The remaining six financial and prohibiting instruments are always specific for groups of processes related to environmentally relevant substances. Their applicability depends on the situations where the flows of these substances occur into, through, and out of the economy. Their design has to take into account the most relevant characteristics of the situation. Financial instruments require an authority that taxes and that collects and refunds deposits. They all require measurable units as the targets to which the taxing rules can be applied. Prohibiting instruments require the measurement of these environmental targets as well, and also the specification of technologies to which the target standards apply as objects.

Which instruments to choose next would thus depend on the case in which they are to be applied. Should all remaining instruments be used and, if not all, how can a selection be made? First, what is "a situation" or "a case"? After a definition of cases is given, the procedure for instrument design will be developed further. The starting point in that procedure is the normative societal preference ordering of the individual instruments as given above. The procedure then invokes the application of the environmental principles, and now for the second time, of the principle of equality. It is a principle-guided design procedure.

What is a case?

It might be assumed that a case of applied policy is defined as one in which a certain instrument is applied. The target of that instrument then is the main constituent in defining the case, at a given administrative level. Thus, in financial instruments in the EC, e.g., with CO_2 as a target, the case is "CO_2 emissions by economic processes in the EC". Now what is "the case" if a problem tax were also considered on all global warming emissions? The case would then become "all global warming emissions in the EC". However, in specifying general design emission standards, one type of product or installation, the object of the instrument, might be taken as "the case", e.g. "large furnaces in the EC". There thus is a self-referential element in "selecting instruments for cases", since the case can be defined only on the basis of the specification of some instrument that has not yet been chosen. An indeterminacy results, since either target or object can be used to define the case. How can this deadlock be resolved?

First, cases such as relatively independent domains of policy could be defined on the basis of only the targets of instruments. The targets of liability and the methodology of LCA, "total damage" and "all problems" respectively, are not relevant as these instruments apply everywhere, not just to "cases". If all problems are combined into a ranked "total problem" there are no cases left. Then environmental policy becomes one "case". This option does not seem relevant, yet. All the other instruments require governmental

[1] These improvements would also be realized by economic instruments, if fully applicable to all problem targets covered in LCA. Ranking of the problems is implied by the set of deposits/taxes. In that case both extended liability and LCA would lose their regulative function.

specification before becoming operational. The target of the remaining most aggregate instrument possible could then be used to define the case. All other instruments would then apply as well, at least to parts of the domain of the case. In fact, the target of the substance deposit, as the most aggregate financial instrument is then the basis for defining the cases. Like the other macro financial instruments it may have "problems" as a target. A first way to define cases is thus to use the context of a single environmental problem, such as "ozone layer depletion".
The second option for case definition is one level of aggregation lower. One environmental problem consists of the sum of all contributions of all relevant interferences, such as substance emissions, to it. The applicability of an instrument, also a problem oriented one, always depends on its applicability to all contributing interferences that cause the problem. Thus, there is also a second possibility to define cases in terms of substances.
Thirdly, there might be possibilities to define cases independently from the object or targets of instruments. One could take cases as essentially arbitrary units. A case then could illustrate how instruments may function "there." Such a case cannot then form the basis of instrument specification and quantification, it is based on their prior specification in another context already.

In Part Five the cases belong to each of the three categories. The case on milk packaging, concerning one complex private decision in society, looks at the working of one macro instrument, the methodology for life cycle analysis, in a quite arbitrarily chosen decision context. No policy design is involved and thus no design strategy is required. The case on energy depletion and global warming is at the problem level, of two problems together, but with main emphasis is on only two substances, CO_2 and CH_4. The case on nitrogen and phosphorus takes these substances as the case, but considers all problems, with the main emphasis on the eutrophication problem. These latter two cases require the application of a design procedure, with a selection and specification of instruments, in one or more rounds.

Both the problem and the substance case have to specify the objects to be regulated by the instruments. This means that all economic processes are to be specified that, directly or indirectly, are causing the interferences (here emissions and extractions) and the problem. The main method of analysis for this specification is the *societal substance flow analysis*. All the processes involved have their empirical peculiarities that may allow, or make impossible, their regulation by one or more of the instruments in the design set.

The flexible response strategy
Given the case practically defined through substance flow analysis, the first question is now which changes are to be effected, which aims and thus priorities are to be set. Principle-guided design does not solve the problem of priorities, it specifies them. In a specific case one might still make a choice to maximize equality, at the expense of environmental quality, or maximize environmental quality, with high costs, little freedom and low equality. Clearly, extreme choices for one principle will involve some sacrifice of the other principles. Ideally, some optimum should be defined. No such optimum choice is available however. As a practical solution, it will be assumed here that some level of environmental quality is set. This may be done in terms of the diminished contribution to a problem or the diminished emission (or extraction) of substances. With an initial situation given, the "end-situation" is then defined. In this very limited way the

environmental principles play their role. The level of desired environmental quality is assumed to have been set in a separate political decision.

The sets of instruments to be compared in the design procedure have now been made equal in the sense that either set realizes the same environmental quality. Secondary constraints could be added, such as on regulative capacity and costs. Regulative capacity, though a serious problem now, is no constraint here, since macro-instruments severely limit the requirements on that capacity. Enough capacity is assumed. The level of costs is the result of the environmental quality aim chosen and the efficiency in realizing it. Setting a specific level of costs thus either has no practical meaning or is incompatible with setting a specified level of environmental quality. No such secondary constraints are applied.

Only the societal principles should guide further choices. These have primarily been condensed into the preference ranking of instruments as given above. This ordered set of instruments is the next element in the design procedure. Only equality has not yet had its full place. It not only applies to individual instruments but may also apply to policies, such as sets of specified instruments for a case. Thus the situation for policy design consists of four elements:

- ◇ the case, as the set of all processes as potential objects of instruments, specified through substance flow analysis (SFA)
- ◇ the allowable level of emission/extraction or problem contribution, a separate political decision
- ◇ the ordered set of macro-instruments, given in the preceding chapter
- ◇ the principle of equality, now applied to a set of instruments, not to the individual instruments concerned.

The flexible response strategy is a procedure for the selection, specification and quantification of instruments in cases. This procedure is case-specific. There is no given set of instruments that can be applied to each case indiscriminately. The design procedure is thus a flexible, case-by-case response to the situation, hence the name *flexible response strategy*[1]. It is similar to a strategy developed for NATO to employ only the minimum numbers and types of military units required to solve the military crisis at hand. It is the opposite of the *massive retaliation* deterrent approach advocated by some environmentalists for environmental policy.

The flexible response design procedure is now very simple. Invariably given in each case is the influence of extended liability and life cycle analysis. If their influence is enough to solve the problem, the procedure stops. However, given the limited applicability of extended liability and the only very high trade-offs in life cycle analysis, this will not usually be the case. The next most preferred instrument is then taken, the substance deposit. Only its applicability is assessed. It will usually cover only part of the flows and will be applied only to them. Then the next instrument in line is applied, the emission tax. It is applied only to the flows not yet brought under the substance deposit scheme, even if it could cover some of the latter flows. Usually, both instruments together will still not cover all flows.

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[1] We have coined the term 'flexible response scenario' in a study for the Environmental Policy Advisory Board to the Dutch government, see van Manen et al. 1991. It distinguished several scenarios.

Then comes the next instrument in line, the estimated emission tax. It is applied only to the remaining emissions, again even if it could be applied more broadly.
This stepwise procedure goes on until all processes in the case have become the object of one instrument, additional to liability and LCA. It then stops. If the list of macro instruments is not long enough to cover all flows, either instruments with a still less macro character could be added, or the inequality of not covering all emissions in a similar manner would have to be accepted.
For the moment the list seems long enough. If all processes are covered, potentially achieving equality, additional instruments would only subtract from equality. In this procedure, the choice of instruments for the case has now been made, based solely on preference ranking and applicability. One peculiarity may be noted. The applicability of the tradable permit requires emission measurement. The emission tax requires the same emission measurement. Since the emission tax is preferred, the tradable permit will never be chosen in the flexible response strategy.

For each step, it is only the applicability or better, the inapplicability, specifying the processes to which an instrument does not apply, that decides whether the next instrument is to be investigated on its applicability to the remaining processes. An example may further indicate how the procedure works. For NO_x from car exhausts, for example, the deposit scheme, number two, does not apply, nor does number three, the emission tax, since continuous measurement at exhausts cannot yet be a practicable instrument for taxing. The estimated emission tax, number four, could apply however. Then the procedure, for the NO_x from car exhausts, would stop. However, if certain cars were excluded, e.g. diesels, for this remaining group the next instrument from the list would be tried, etc. Number five, general installation permits, is not applicable either. Number six, the general design emission standard, is fully applicable to diesel vehicles. Thus the strategy is flexible, responding to the situations where the instruments are to be applied. Instrument choice involves increasingly less aggregate instruments for the ever smaller number of remaining processes not yet covered.

The instruments for the case now having been specified, the next question is at which quantitative level they should be set. Instrument quantification is to result in the allowable level of emissions or problem contribution. An economic model of the case situation is the basis for this quantification and equality its guiding principle. The trade-off is set at the level indicated by the model. Less sophisticated models may require an iterative procedure[1]. The trade-off is easily implemented in substance deposit, emission tax, and estimated emission tax. The level of the deposit and the tax are simply set equal to the minimal trade-off required. For the prohibiting instruments, the trade-off is determined implicitly, when setting the standards. The procedure for setting design standards should thus take into account that trade-off, to arrive at standards where the estimated costs per unit of emission / problem reduction are more or less equal to that trade-off. Such procedures occur seldom now, since there is no explicit trade-off to refer to. If no level of the trade-off could realise the environmental aim set, a highly improbable situation, then further instruments could be <u>added</u> to improve its efficacy.

Purists might object that the empirical relationship between a quantified level of instruments and the desired level of environmental quality is extremely complicated,

[1] Since the situation in the case is dynamic, there will not be only one level of emission result for example, but a time series. The environmental aim could be defined for a certain year.

mainly based on long-term dynamic effects, and variable over the course of time. They are right. Empirically substantiated quantified models will not usually be reliable or will not be available at all. Thus this part of the design procedure cannot be very rigid, as is true for all design processes. However, an approximating procedure could be followed that would approach the required level of trade-off quite reasonably. That approach would be to make a survey of the techniques for emission reduction that could be applied, now; to rank them according to the increasing level of expected costs per unit of emission reduction; and to assess which at least is required to realize the stated environmental quality aim for the case[1]. Its costs per unit of environmental improvement then are taken as the value for the trade-off. If this rough quantification of the trade-off is the starting point, each instrument may be quantified when introduced. It will eventually become clear whether that level has been set too low, or overshoots the environmental mark. There is good reason to expect that changes in the level of the trade-off will be required only rarely. Dynamic technological effects will decrease the environmental interferences in the course of time. Increased production will somewhat offset this reduction. A lowering of the level of the trade-off and the tax would then be due. With increased welfare the value of environmental quality increases, however, requiring an increase in taxes since the level of environmental quality would be set higher. If the initial choice is based on the costs of the then available technologies, it might be a stable choice for decades.

3.4.5 Conclusions

The design of macro environmental policies through the flexible response strategy is the result of five steps.

First, the three social principles of efficiency, equality and freedom specified in the preceding chapter, are applied to the most macro types of the single instruments available. Three rankings result. The environmental principles do not apply at the level of individual instruments, but only at policies. Efficiency should be made operational only as dynamic efficiency. Freedom is specified as the lack of impediments, per instrument. Equality is the only social principle to apply at the level of policies, as sets of instruments, together with the environmental principles.

Secondly, the three rankings of instruments based on societal principles are combined into one. The not-yet-developed improvement eco-audit falls off. Extended liability is included, not because of its attractiveness, but because its continued development seems unavoidable. The resulting ordering follows the familiar sequence of instruments, from cultural, to financial I and II, and to prohibiting I and II, with the structural instrument not explicitly ranked.

Thirdly, the cases are described. Cases are defined in terms of one problem, including all relevant substances, or as relating to one substance only, in its relation to all environmental problems. The design case is described empirically in terms of the material economic analysis of the substance flows concerned, using the Substance Flow Analysis method.

Policy design at the case level starts with the choice of instruments, the fourth step. Starting at the most preferred instrument, instruments are added till all flows are covered

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[1] As indicated above, there will not be much reason to expect a change in the level of taxes. More probably, the environmental quality aim will become more stringent as welfare increases and costs per unit of emission reduction go down.

by one of the case-specific instruments, a first step towards equality. No more instruments are added since they would lead to inequality. This procedure at the same time specifies the objects and the targets of all instruments concerned. This design procedure is adaptable to the situation encountered, the whole procedure has therefore been named the *flexible response strategy*.

The fifth and last step in the flexible response procedure is to quantify the instruments. The quantification concerns the choice of the minimum trade-off that is required to arrive at the environmental aim set. The trade-off should be set equal in all instruments, both for reasons of efficiency and equality. With financial instruments, that trade-off is given explicitly in the level of the deposit or tax. With prohibiting instruments, an explicit bureaucratic procedure, not now generally used, may approximate the equality in trade-off. A short cut method may arrive at a reasonable quantification. Changes in the level of the trade-off chosen will be required only very rarely.

3.5 CONCLUSIONS ON PRINCIPLES AND STRATEGY

Part Three has seen the development of a design strategy for instrument application, the flexible response strategy. The steps leading to this strategy start with a discussion of the way principles may be used in policy design and a choice is offered on how to use them in environmental policy. The choice is between an ultra-utilitarian approach, where the desirability of instruments is assessed at the case level only, and the *principle-guided choice* of instruments, where an order of preference is established on the basis of general notions, regardless of specific cases. In both cases, the content of the principles could be the same. The principle-guided choice has been selected as the more rational and more practical alternative.

The principles involved are those of political philosophy, indicating which institutions for society are just. Three societal principles have been chosen as the most relevant for policy design in environmental policy and have been specified for that domain of application:

- ◇ *efficiency* the lowest social costs per unit of environmental improvements
- ◇ *equality* an equal trade-off between environmental improvements and private costs for all regulatees
- ◇ *freedom* the lack of infringements by government on private choices.

Environmental principles were specified in the discussion on targets of instruments in the preceding Part Two. They may relate ultimately to three main value areas, i.e. human health, quality of nature, and functions of the environment for man. More operationally, they may be expressed as problems caused by human activities, such as global warming, ozone layer depletion, acidification, etc. or per type of interference, e.g. the emission of a substance, related to all problems concerned. Environmental principles play only a limited role in the design process, in helping define cases for policy design and setting the aims of environmental policy. A quantitative choice of environmental aims is assumed to be made in a separate political decision, not discussed here.

The societal principles may first be applied to individual instruments. This results in a priority list of macro instruments, on which all less-macro instruments score lower for the three principles. The resulting order is

1. Methodology for LCA
2. Substance deposit
3. Uniform emission tax
4. Estimated emission tax
5. Uniform tradable emission permit
6. General emissions standard
7. Estimated product standard, LCA
8. Estimated emissions standard.

Extended liability, though scoring better than direct interventions, cannot be ordered easily and is extremely insecure in its general evaluation because of unpleasant side effects. No ranking is given.

Extended liability and the methodology for LCA apply regardless of cases. The other instruments require specification in cases. For substance and problem cases the procedure is a flexible, to start with the most preferred instrument and to apply it to all processes where it is reasonably applicable. For the not yet covered processes the next instrument in line is applied, etc. When all or virtually all processes in the case are covered, the quantitative level if the instrument is set at a level that the environmental aim will be realised. This design procedure is the ***flexible response strategy***.

In this design procedure a set of instruments is being developed that realises the highest expected efficiency, leaves most freedom to the citizen in avoiding infringements on their choices and treats everybody in all his decisions as equal as possible, while at the same time being effective environmentally. Thus, equality, freedom and efficiency are maximized for a given level of environmental quality.

This section on principles concludes with a few remarks about the relation with one other well-known principle for environmental policy, the polluter pays principle. The current polluter pays principle requires that polluters pay their own costs of emission reduction[1]. If standards, the main prohibiting instrument now, have been set the costs involved should be paid by those directly regulated. These may pass on the costs to others, if the market allows them. A change from current, mainly prohibiting instruments towards the macro instruments of the flexible response strategy would imply a breach with the current principle. The financial instruments dominant in that strategy make the polluter also pay for all damages to collective goods, current and future. The rules for private damages and collective damages, now diverging, would unite. All damages to third parties, collective or otherwise, would be treated the same by holding the party causing them financially responsible through some form of liability. For private type damages there is private liability, administered through courts. Administrative rules as in emission taxes and substance deposit, would bring about liability for the mostly collective environmental damages. Thus the right to not being disturbed by others, protected by liability, would be extended to the environmental domain as well.

[1] The principle has been stated in "Recommendations of the Council on Guiding Principles Concerning International Economic Aspects of Environmental Policies", OECD, 26-5-1972 and the "European Community's Programme of Action Concerning the Environment", Publication no. C 112, p. 1 ff., 20-12-1973. The principle has subsequently been specified and revised, with the addition of provisions for exceptions, in "Recommendation of the Council on the Implementation of the Polluter-Pays Principle" OECD, 14-11-1974, and for the European Community in "Recommendation of the Council Concerning the Attribution of Costs and the Role of the Government in Environmental Matters", Publication no. L 194, p. 1 ff., 25-7-1975. On 1 July 1987 the principle became binding for EEC environmental policy in the Single European Act, Art. 130R, Section 2. See Huppes 1989 for a further analysis.

PART 4 INSTRUMENT DESIGN
DETAILED DESIGN OF MACRO INSTRUMENTS

4.1. INTRODUCTION TO THE DETAILED DESIGN OF TWO MACRO INSTRUMENTS

* (Extended liability)
*** Methodology for LCA**
*** Substance deposit**
* Uniform emission tax
* Estimated emission tax
* Uniform tradable emission permit
* General emissions standard
* Estimated product standard, LCA
* Estimated emissions standard

In this section the design of the two most preferred macro instruments will be worked out in more detail, the *standard methodology for the life cycle analysis of products* and the *substance deposit*. Like all other instruments for environmental policy, the two macro instruments investigated here consist of two interfaces. Introducing more detail in the design thus involves working out the two interfaces in more detail. This will be done for both instruments. Extended liability will not be treated further, as has been indicated in the preceding chapter. The other instruments on the list have been worked out in more detail already, both in the literature and in practice. Being less macro, they also are simpler in their analysis.

The basic definitions of the designs have available been given. In Part Two, two promising macro interfaces linking society to the environment were identified, as combinations of environmental targets and economic objects. The first such interface is the problem oriented life cycle analysis of products (LCA) and the second is the substance flow analysis of all economic processes (SFA). With these interfaces two macro instruments were constructed by adding to each a working mechanism, also of a maximally macro character. A cultural mechanism added to LCA resulted in the standard methodology for life cycle analysis. An economic mechanism added to SFA resulted in the substance deposit system. These two new macro instruments are the most preferred instruments in the strategy for macro-environmental policy design developed in the preceding section. Both instruments have been developed at a general level only and need further elaboration. Contributing some elements to this further practical development is the aim of this section.

The approach followed here is the same for both instruments. First, their analysis at the society-environment interface will be described in more detail. Especially in the case of the life cycle analysis, this involves quite complicated reasoning. "The processes required for the functioning of a product", as the object of the interface, is not some fact available in a database. Since actual processes usually function for different products, these actual processes have to be dissected into analytic parts that contribute to the product investigated. The social working mechanism is then added to both LCA and SFA, also in greater detail. Possibilities for further aggregation will be examined for both instruments.

The more detailed designs given in the next two chapters build on collective work carried out at CML during recent years, especially in the Substances and Products Section.

Certain publications summarizing that work give more detailed information on many more subjects than are treated here.
In chapter 2, the standard methodology for life cycle assessment is defined in more detail. The description of the analytic method is based on Guinée et al. (1993; 1993b). The most specific study on methods is Heijungs et al. (1992), the result of an extensive project on life cycle analysis by CML for the Dutch central government. See also the general survey on LCA in Assies (1992a). Some theoretical elements are added here, that have first published on in a paper for a SETAC Workshop in 1991 (Huppes 1992) and in a paper for a CESIO conference in 1992 (Huppes and Guinée 1992b). The themes are:

◇ how to structure the data on the processes required for the functioning of one product (format, section 4.2.3)
◇ the attribution of process parts to the product analyzed, if a that process contributes to several products (allocation, section 4.2.4)
◇ principles for the general set-up of the environmental problem analysis (classification outline, section 4.2.5)
◇ principles for the aggregation of several problems into the "total environmental problem" (ranking problems, section 4.2.6)

Finally, the social working mechanism at the government-society interface is worked out here, in section 4.2.7. It is contrasted with less aggregate alternatives, such as systems of ecolabelling. Some attention will also be directed to possibilities for further aggregation of the object. From the function of one product, the object may comprise?] more complex activities implied in "food" or "recreation". The next step could be to the still more abstract unit of "the function of an ECU spent on consumer goods".

In the next chapter the substance deposit is worked out as an economic instrument with its society-environment interface defined in terms of the substance flow analysis. The description of the method of substance flow analysis is primarily based on van der Voet et al. (1992a). Next, the financial working mechanism is added, resulting in the substance deposit. The main lines here agree with those given in Huppes et al. (1992). A central question at this interface is at which administrative level the substance deposit may best be applied. Given a certain administrative level, further aggregation can only take place through aggregation of the target, since the object already encompasses all processes involved in society. The first step towards such a further aggregation discussed here is that from substances to problems.

The next two chapters thus each treat one instrument, first its society-environment interface and then its government-society interface. The relation between the two instruments is discussed in the concluding chapter 4. Are they complementary, mutually exclusive, or only partly overlapping?

4.2 STANDARD METHODOLOGY FOR LIFE CYCLE ANALYSIS

4.2.1 Introduction

The first main instrument selected for macro environmental policy is the standard methodology for life cycle analysis. It has a cultural mechanism at the government-society interface. The life cycle analysis is used to make the society-environment interface operational here, as the substance flow analysis is used to make that interface operational for the substance deposit, in the next chapter, 4.3.

Life Cycle Analysis (or Assessment, both mean LCA) has been developed as a decision-support system. It helps to consider the environmental effects in decisions on products. LCA developed in the Sixties and Seventies as a private tool for analysis, not as part of an instrument for public policy[1]. In the Seventies and Eighties many LCAs were made, mainly by and for private companies in the USA (e.g. Hunt 1974) and some in Sweden (e.g. Sundström 1971) and Switzerland (e.g. Basler and Hoffman 1974). In the Eighties a broader interest emerged in Europe. The Swiss central government initiated some studies (EMPA 1984) and started data collection in several studies on basic material production processes (by Thalmann, Humbel and Fecker, see Guinée et al. 1993 for references). The Dutch government commissioned the first study on methodology (Druijff 1984) and several case studies. By then, a number of problems had emerged that have not yet been solved. These problems relate to very different aspects of the life cycle analysis. Many basic questions have not yet been answered.

◇ What is the aim of the analysis? In the US the emphasis is on product improvement, in Europe on product comparison.

◇ Which data are relevant? Data on past, current or future processes; data on average, best or worst processes; data representative of local, regional or global processes, to name but three often unclarified dimensions.

◇ How are multiple processes to be treated? For example, to which of its several products do the emissions of a given refinery belong? To which of the different waste streams burned are the emissions of a waste incinerator to be attributed?

◇ Which environmental data are to be included, related to which environmental effects. Do workers' health and safety belong to LCA (Danish studies), or user safety, or resource use? Or should the analysis cover the full range of all relevant aspects? In which case one could include costs, competitiveness, and different types of labour requirements, as some Germans do (e.g. öko-Institut 1987).

◇ How are the fast growing number of environmental interferences to be processed into meaningful units? In the US emphasis is on adding different types of resource use and emissions by their mass (see Hunt 1991 and Hocking 1991). In Europe, the emphasis is on specifying separate effects in the environment, under such major problem headings as resource depletion, health risks and acidification. This latter approach is in line with the problem oriented definition of targets stated here in the preceding section.

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[1] Pioneers in the subject are Sundström in Sweden, Hunt in the US, and Basler and Hoffman in Switzerland. An early LCA-like study was ordered by Coca Cola as early as 1969 to be conducted by the Midwest Research Institute. Franklin Associates became an independent offspring of that institute and Hunt one of the associates. See Assies 1992c for these references and for a historical survey.

◇ How are products with different scores in different directions to be evaluated? Is one kilogram of SO_2 emitted into the air as serious as one kilogram of cadmium in the sea? Is energy depletion as important as acidification, and in what units does one measure that importance?

In the Eighties it became clear that if life cycle analysis was to become broadly used, the basis for answering such questions should be established in a more systematic manner. Two types of activity developed. The first is primarily social, with an emphasis on procedures and terminology. A common procedure may eventually be agreed by practitioners in a Code of Practice. The widespread confusion on terminology might be alleviated through the development of a common framework for analysis. Many elements of confusion still exist. Does 'raw materials', as one type of input into a process, refer to materials that have been extracted from the environment already, as a product of the extraction process (Vigon et al. 1993, p.43) or as the resource that is extracted from the environment (this study)?
The other type of activity is primarily scientific, to work out the methodology of LCA in more detail and make it operational in a general manner, as opposed to an *ad hoc*, case-specific manner. The former activities are mainly collective and procedural and related to the influence and power of the parties involved. The latter activities mainly consist of individual or small group scientific activities, with their dynamics related more to scientific criteria.

In the last few years all types of developments in LCA have accelerated, with a central international role for SETAC (Society for Environmental Toxicology and Chemistry), including SETAC-America and SETAC-Europe, closely linked but not fully integrated sub-organizations. Until recently at least, the emphasis in the US was on converging procedures, related to framework and terminology, and in Europe on the development of theories and methods. The American working method is the workshop, collectively producing results in a social process. In Europe, the emphasis has been on constructing methods, or theories that may partly converge, but also diverge. This is not a collective endeavour. It is individuals and single-minded groups that build theories, although obviously not in a social void. In the SETAC-Europe workshops, theories on several topics have been discussed and published under the name of the individuals and groups that stand for them (e.g. SETAC-Europe 1992)[1].

Further distinctions associated with these two types of activities indicate at least five levels of analysis, each with its own impetus and aims, divergence and convergence. In increasing order of specificity, the following levels can be distinguished[2]:

1* Code of practice / general framework
2* Main options for parts of methods, general
3* Specific LCA methods, mutually exclusive
4* Tools for practical application of different methods
5* Applications of different methods in cases

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1 These different aims in different SETAC workshops have led to much confusion.
2 This treatment owes to a discussion with C. Pesso, from OECD Paris.

At a SETAC meeting on the Code of Practice, in Sesimbra, Portugal in April 1993, there seemed to emerge a consensus on a code of practice and a general framework. This consensus was driven by the desire of practitioners and firms using LCAs to arrive at uniform results. That zeal, however, resulted in main method options and even certain parts of very specific methods, such as allocation, being included in that draft code of practice[1]. Contrasted to this convergence is the partly diverging scientific development of the specific methods required to execute LCAs. The tools for practical application, such as software and databases, also show a corresponding divergence. In certain cases, these differences in methods and tools may thus lead to substantial differences in results, see Ekvall (1992) and Guinée et al. (1992b). Several parts of the method of analysis are now being set up in more detail, both through the work of individual scientists and through the forum organizing activities, especially those of SETAC. Increasingly, other supranational and international organisations are also becoming involved, such as UNEP, OECD, ISO and the EC. With newer methods and tools being developed, already existing divergence may even increase for some time to come.

The methods that have been developed not only differ one to the other. They also do not fit neatly into the general framework provisionally described in Portugal. Heijungs et al. 1992b, in the foreword to their English edition (written in 1993), summarize the differences between the framework of their method and the framework of the SETAC Portugal Code of Practice as given below, see table 4.2.1. The framework in their report consists of five components. The draft Code of Practice consists of four components. The main difference concerns the *classification* and *evaluation* components. Both are part of the *impact assessment* in the SETAC framework. *Classification*, as used in their report, is subdivided into *classification* and *characterization* in the Code of Practice, where the former denotes the labelling of interferences according to the effect categories they may contribute to, and the latter amounts to actually computing the contributions to these different effect categories[2]. See the survey of the similarities and differences between the two approaches in the table below.

In this study the term *impact* has been avoided mainly because it covers both environmental interference (= intervention in Heijungs et al.) and the effects of this in the environment. *Interference* here indicate human intervention in the environment, e.g. resource extraction and emissions (environmental releases). *Effects in the environment*, as indicated in the environmental profile in the classification, mean the resulting environmental problems, e.g. resource depletion, global warming and acidification.

Also not fully in line with the draft SETAC framework are Keoleian and Menerey (1993), who in a study for the US EPA conclude that "*the final result of the impact analysis is an environmental profile of the product system*" (p.108), while the provisional SETAC framework of Sesimbra ends the impact analysis with a "valuation".

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[1] It is still at a preliminary phase, to be discussed further in sessions in the Europe and the US and finalized, in a procedure still to be determined (May 1993).

[2] This distinction between *classification* and *characterization* hardly seems significant enough to be specified in a Code of Practice.

TABLE 4.2.1 THE GENERAL FRAMEWORK FOR LCA IN THE DRAFT CODE OF PRACTICE IN SESIMBRA 1993 AND IN THE LCA GUIDE OF HEIJUNGS ET AL. 1992.

	Code of Practice Sesimbra - April 1993			LCA GUIDE Heijungs et al. October 1992
1.	goal definition and scoping			1. goal definition
2.	inventory analysis			2. inventory analysis
3.	impact assessment	3a.	classification	3. classification
		3b.	characterization	
		3c.	valuation	4. evaluation
4.	improvement assessment			5. improvement analysis

In the following sections some theoretical contributions are made, now divergent, but as much as possible related to the general SETAC framework that seems to be emerging. Of course it would a good thing if the theories themselves converge into one common framework. However, for this author, as one of the originators of the Dutch programme that resulted in Heijungs et al. (1992b) and as a participant of the SETAC discussions, it seems too early for theory convergence. That would block the discussions on theory that have just started. Obviously the aim is that what is constructed here are parts of the "right" theory, so that what may seem divergent now may become the convergence of tomorrow.

First to be discussed are possible applications of LCA, that constitute the reason for executing them. Then the CML/TNO/B&G framework (that is Heijungs et al. 1992) is briefly described, as it has been developed in the LCA methodology project for NOH/NOVEM, a combined Research Programme of the Dutch Ministries for Economic Affairs and the Environment. This section on the framework and applications, 4.2.2, may be read as a technical introduction to some elements of an LCA theory, following in sections 4.2.3 to 4.2.6. An initial element, based on Huppes (1992) and Heijungs et al. (1992b), is how to describe economic processes, as the basic units of information for the inventory of LCA. It is the *knowledge format* of the content of that information that is discussed in section 4.2.3. Only in exceptional cases will the processes required for the functioning of a product be dedicated to producing for that product alone. They usually serve several functions in different ways. Refineries, for example, have one flow of emissions but a widely diverging stream of products. For such a multiple process, the environmental interferences must be allocated so that one fraction of them goes to each of the products produced. In section 4.2.4, the problem of *allocation* occurring in all multiple processes is treated. The solution proposed there is not yet fully operational. It is certainly not yet a point of international convergence. This treatment of allocation also is a much revised version of Huppes (1992). This is followed by a description of the general *classification*, in line with the problem oriented target description developed in Part Two, see section 4.2.5. Finally, as a last item on LCA theory, a most-preferred-method for the *evaluation* of the relative scores of products is worked out, in section 4.2.6. The latter two subjects build mainly on Huppes and Guinée (1992) and Huppes and Heijungs 1993.

These subjects on LCA theory form a selection only. The fullest treatment on general LCA theories is in Guinée et al. (1993 and 1993b). The most detailed methods for executing LCAs currently available are in Heijungs et al. (1992b) produced for the Dutch government, Nordic Council (1992), and in Keoleian and Menerey (1993), produced for the US government. In a companion volume to the method specification, the Backgrounds document, Heijungs et al. give an extensive survey of the parts of LCA theory that are disputable or have not yet been sufficiently worked out.

Finally, the second and very much shorter step of the instrument design, the cultural working mechanism is added, in section 4.2.7. The chapter ends with conclusions. Sections 4.2.3 to 4.2.6 are quite technical. Readers less interested in the technicalities of life cycle analysis may wish to skip these sections.

4.2.2 The framework of LCA

Relation to instrument design

The interface of the problem oriented life cycle analysis of products has been defined here in terms of object and target, a terminology not now used in LCA discussions. The object is the life cycle of a product in the economy, consisting of the processes required for the functioning of that product. That is the product system, a segment of the total economy. The target level may first be that of environmental problems or that of "total environmental problem". These two elements of the interface, objects and targets, have a corresponding place in separate components or elements in the framework for LCA developing. The economic object is analyzed in (part of) the *goal definition* and in the *inventory*. The aggregation of targets has its place, first, in the *classification*. In many cases the classification of two products leads to conflicting scores in terms of problem contributions. Their comparative evaluation would then require the weighing of different problems. As no modelling is available, this weighing is essentially a normative evaluation. All such evaluations are to be made in a separate component, or a separate part of it, the *evaluation*[1]. A formalized evaluation would establish a further integrated target, as the most macro target currently possible. See figure 4.2.1 for the components of LCA that are relevant, partly or fully, for instrument design here.

The nature of LCA in applications

The general LCA instrument envisaged should be as independent from specific applications as possible, in order to cover them all. Its kernel is the common denominator of all these possible applications. The application itself is no longer part of the environmental LCA. Several applications are possible, see the list in figure 4.2.1. Ultimately, all these applications require the environmental comparison of two or more product systems. The major groups involved are consumers, including NGOs, manufacturers and the public sector. The consumer selection of products implies a comparison between existing product systems. Producers may supply that type of comparative information externally, to consumers, to other firms, or to public bodies, or only internally, as in the design of new products. The destination of the information does not alter its nature. Moreover, such internal application as product design requires a comparison with one or several imaginary product systems considered, usually specified

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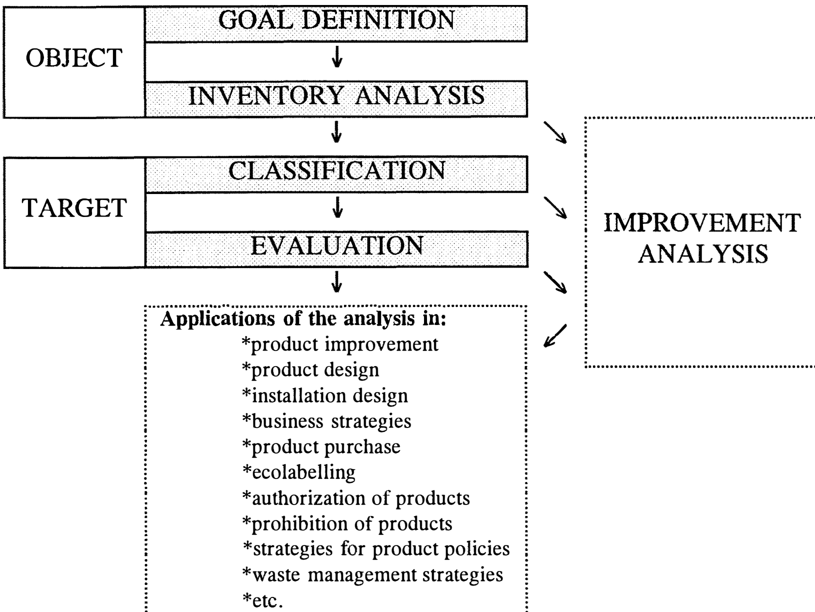
[1] In the draft SETAC Code of Practice, the evaluation has been termed *valuation* in LCA, while the life cycle analysis, as the decision support tool, is called life cycle *assessment*. This terminology is different from that used in other relevant contexts, especially that of decision theory. An authoritative textbook on decision analysis, such as Bell et al. 1988, does not even have the terms valuation and assessment in its subject index.

FIGURE 4.2.1 MAIN COMPONENTS OR ELEMENTS OF THE LIFE CYCLE ANALYSIS OF PRODUCTS AS RELATED TO INSTRUMENT DESIGN AND APPLICATIONS



on paper only, or perhaps taking the form of a prototype. The governmental development of policies for specific products also involves a comparison between product systems, as in ecolabelling. Forbidding a product or allowing it on the market requires a comparison with the environmental effects of some real or an imaginary "standard product system". More exotic and complex applications that may develop, such as formulating business and policy strategies also require comparisons. These are between larger numbers of systems of yet unspecified product designs. Other applications that might evolve, such as determining alternative techniques to reduce emissions, also require an environmental comparison between alternatives. A specific application of this type, now investigated by the EC, is the support of decisions on waste treatment systems.

One application has received the greatest attention in framework discussions until now. It is the environmental redesign of existing product systems, or *product improvement*. It is intricately linked with the improvement assessment of the Sesimbra Code of Practice. One possible reason for its prominent place in the framework is that in the US explicit comparisons, though made, cannot be used in advertising because of the risk of prosecution, as the federal prosecutor has indicated. The improvement assessment specifies attractive options for improvement[1]. This component has an ambiguous status since it is an already complex step in a very specific application, product system improvement. See De Groot (1992a, chapter 3) and Keeney (1988) for a general treatment

[1] Contrary to the improvement assessment in the draft Code of Practice, the improvement analysis specified by Heijungs et al. does not require the generation of options. It specifies where changes in the product system would lead to the greatest changes in desired results, thus indicating only where to look for the potentially best options.

of value-based options generation, and more specifically Heijungs et al. 1992b, p.58, figure 5.2 on the LCA-based product innovation procedure. Option analysis is not yet the specification of alternatives in the goal definition followed by their environmental comparison. That would be the usual LCA comparison made for any application, and therefore does not require a special status in the general framework. It is not alternatives that are assessed, but the options with which alternative product systems may be designed, eventually to be compared in the next round of LCA. Any reasonable selection of design options seems to be based on the following:

◇ technical options regarding the design of the product and product components (as parts) that would be functionally and economically feasible
◇ technological (= technical + social) options for process improvements that are economically and functionally feasible
◇ the introduction of new processes, e.g. the separate collection of one or more types of waste from the product system to allow their recycling.

Such design-supporting activities seem most sensible, not only for the improvement of existing products, but also for new designs of products and processes. They are, however, part of a much broader analysis than that restricted to LCA, requiring technical, economic and functional data, and the capacity of designers to combine these in choices. In the assessment of options, e.g. evaluating improvement possibilities or design options, an evaluating comparison is always involved. In the framework as required for the instrument construction there is no place for the improvement analysis or the improvement assessment. What is required in all applications, also in product improvement, is the comparison between product alternatives and product variants.

Thus all applications require a comparison between alternatives, existing, imaginary, very similar or very different. The general goal of any life cycle analysis, in a technical sense, may thus be defined as the *environmental comparison of two or more product systems*. The first four components in the Dutch LCA guide or at least main parts of them are required for that comparison. It is the same elements that are required for the specification of the interface. The four components relevant to instrument design will now be briefly described, together with an indication of which elements are relevant to the policy instrument or may be left unspecified in this context. The description is detailed in the steps specified by Heijungs et al. 1992b.

The framework of the Dutch LCA guide
The Dutch LCA guide specifies four components relevant here, goal definition, inventory analysis, classification and evaluation. Each component has been detailed in a number of steps.

Goal definition
Step 1 Determining the application
Step 2 Determining the depth of the study
Step 3 Defining the subject of the study

The goal definition indicates the products to be compared and the basis of their comparison. The first step, the specification of the intended applications, is not relevant to the instrument, since ideally it covers all applications. In a technical sense all LCA are a comparison of product systems. The depth of the study, closely related to the intended application, is not a relevant characteristic here either. The type of application and depth do not influence the structure of object and target. In defining the subject of the study, the third step, several elements are of paramount importance. Which products, from which product group, are to be compared, functioning where and when. For this comparison a unit is required in terms of which the products to be compared are equal. That unit is the *functional unit* of the products analysed. "Drinking one litre of milk" is an example. Without this functional unit, even the quantification of a single product system is impossible. The functional unit must be specified in terms of space (e.g. drinking milk in the Netherlands) and time (e.g. in the Nineties). The ways the functional unit may be realized, in specific product systems, should also be indicated (e.g. the transport of packaging made of PE coated carton, or 420 gramme returnable glass bottles, or 70 gramme returnable polycarbonate bottles) they are the products, or better product systems, to be compared.

Inventory analysis
Step 1 Drawing up the process tree
Step 2 Entering the process data
Step 3 Application of the allocation rules
Step 4 Creating the inventory table

The inventory analysis supplies the data on all environmental interferences caused by one functional unit, for each of the product systems to be compared. The specification of the process tree, step one, shows which processes are relevant and how they are related. One branch of the process tree for polycarbonate milk bottle, for example, is all the processes required for the production of the PE lid of the bottle. These range from oil extraction to the transport of the manufactured lids to the dairy. Specification of the boundaries of the product system with the environment is one main element in this step. It would appear to be a waste of time and the cause of much unclearness in results if this step were made anew in each case study. The boundary is best fixed beforehand in a well defined method so it is not variable according to case, at least not in most the most common processes at the boundary like forestation and final waste handling.

The next step is entering the data of all processes involved. Examples are the data on oil refinery for the PE lid and the recycling process of the dirty PE caps. Here the way the data are to be structured is a major problem, since it should allow any desired subsequent detailed analysis. The specification of data should be independent of the specific LCA application. The method of data structuring and the type of data included, both related to the knowledge format, should not be variable. As with the boundary problem they should be specified beforehand in the method used. The subject of the data format is treated below, in section 4.2.3.

Many of these processes relate to more than one product system, as in the refinery example. Only a part of such a process, caused by, or required for the realization of the functional unit of the one product analysed, is to be allocated to that product system. The allocation method specifies a measure of the relative contribution to the full process. As with system boundaries and format, the choice of allocation method should not be made at the case level. If possible, it is specified in advance, in the method to apply. It is a central element in the object definition of the instrument. The different types of allocation problems involved, and the preferably uniform method to apply to them, are the subjects of section 4.2.4.

Creating the inventory table, the last step of the inventory, involves the quantification of all the allocated processes caused by one functional unit, and the addition of all their environmental interferences. This addition may proceed stepwise, into increasingly larger units of groups of processes, which together form a process again (e.g branches like product components, or stages in the life cycle). The stepwise procedure allows an analysis of which types of processes contribute most to the environmental effects. The final addition depicts the product system as one single process. That process delivers its function and in doing so causes the environmental interferences given in the inventory table.

Classification
Step 1 Selection of problem types
Step 2 Definition of classification factors
Step 3 Creation of environmental profile
Step 4 Normalization of effect scores

The classification translates each interference into a contribution to a number of environmental problems. The first step is the choice and exact description of the environmental problems involved in the analysis. This step, again, is not to be specified at the case level. It is a central element in the specification of the target of the policy instrument, together with the modelling of what the contribution of a substance to each problem is quantitatively. The LCA guide specifies both provisionally and certainly not as a finalized state of the art. Further observations on both steps will follow in section 2.4.5. These problem definitions will not be worked out to an operational level. Given the specification of these first two steps, the third step, the computation of the environmental profile or ecoprofile, is primarily computational.

In the next step the scores in the ecoprofile can be normalized into "units of problem contribution". The global warming contribution by all interferences of the product system is a percentage of the total global warming contribution of all global warming substances emitted on earth together in a certain year (it has the dimension [year]). The same may be done for ozone depletion, acidification and any other problem specified. The result would be a normalized environmental profile, indicating how high (not how important) the relative contribution of the product system is to each problem. Drinking one litre of milk from a polycarbonate milk bottle might contribute $12*10^{-12}$ year percent to global warming; $3*10^{-12}$ year percent to ozone layer depletion; $47*10^{-12}$ year percent to acidification; etc. (fictitious data).

Evaluation
Step 1 Evaluation of the environmental profiles
Step 2 Evaluation of reliability and validity

The evaluation is the normative comparison of the environmental effects, in terms of problems, contributed to by the different (variants of) product systems. If one product is better in all respects than another, it is to be preferred. That alternative is therefore dominant. This will hardly ever be the case, however, certainly not at the level of the interference table and only seldom at the level of the ecoprofile. Then the contrasting scores must be weighed or no total evaluation will result. This weighing can be done ad hoc, on a case-to-case basis. For the design of macro instruments, that option is not relevant. Or the weighing can be formalized, by assigning a weight to each problem in terms of its relative importance. Section 4.2.6 deals with the formalized ranking procedure.

Step 2, the evaluation of reliability and validity, highly important in case studies, is not an element of the interface, a combination of object and target.

Short-cut methods for instrument design?
One question for instrument design now is whether all these complicated LCA elements or components should be part of it. Could a general indication of this interface suffice, or are there simpler methods available, such as fast, short-cut, screening? The answer which is no, will be briefly discussed here. Nevertheless, such short-cut methods could be essential to many practical applications of LCA, especially in product design and product improvement.

The full LCA specified here will usually be a complex and expensive affair. There are two ways for simplifying the analysis. One is based on the full method, but simplifies it for specific applications and supports it with time-saving tools. The other is to replace the method with a simpler one.
First, in some applications the comparative LCA may be extremely simple. In a design procedure for a product, some part of it may be made lighter but strong enough, from the same material. The only difference between the two variants then is the amount of the material used. If that material has negative environmental effects, the evaluation gives a perfectly clear result. No further quantification is required and no weighing of problems is necessary. The framework for this evaluation is still the full LCA analysis. This simplification first results from leaving out all components or process branches of the product systems that are the same in both variants. Only a difference analysis is needed. Design improvements often have this result. The second element in the simplification is that the one product is dominant over the other at all interferences, hence in all problem contributions, and hence in its evaluation, regardless of the specific interferences.

Simplification may also result from the omission of all processes that are relatively unimportant, prima facie. The check then is the full application. A simplification of the practical work may also result from tools that make case studies easier. Suitable databases on main processes, software for all types of computation, and software for guiding the LCA procedure itself can together increase productivity in this field by orders of magnitude. All these simplifications still require the full method at the conceptual level, and thus in the instrument.

Secondly, the only real alternative to LCA is the use of a different and simpler method. At perhaps a higher level of abstraction, however, that method would still have to predict the results of the full LCA. No "other theories" are available. One attractive option would be the use of a simple but powerful indicator for the results of the full LCA. A first indicator could be economic costs. The more expensive a product is, the greater its

requirements on different material processes will usually be. As a first proxy this may be a reasonable guess. On average, that indicator must be right, since all products together cause all environmental effects. The interesting thing about LCA, however, is to determine the nature of the variations not reflected in prices. Indicators have been sought, especially for guiding product design, see van Weenen (1990) for a survey. None of the indicators is full proof, since counter examples exist in more quantified analyses. Recyclability is one of the criteria most often mentioned. The first Swiss studies on milk packaging (EMPA 1984) already indicated that to the best knowledge then available the throw-away polythene bag was environmentally preferable. More recent research by Migros, who used newer data and a different method of classification and evaluation, still singles out the throw-away polythene bag as environmentally best. Generally valid indicators do not exist now. The more general point here is that such indicator-based and often more qualitative approaches, should be tested with several applications of the full method. In the final analysis the full method remains the test; there is no independent "other method." If a policy instrument is established as a combination of the full LCA interface with a working mechanism, that working mechanism will also work towards the development of simpler, tested methods. This highly important subject in a practical sense is thus not pertinent to the design of LCA-based macro policy instruments. Its development will be stimulated by the general LCA instrument.
The four more technical subjects on LCA theory indicated above will now be treated in the subsequent sections.

4.2.3 Inventory: Format[1]

A product system consists of individual economic processes, that together constitute the process tree. Such processes may be defined at an aggregate level ("PVC production"), but then no data is available. They may be defined at a minute level of pipes, and nuts and bolts, but no data is available there either. The right level is the one where relevant data are to be found, see in this sense Fava et al. 1991. Which types of data are relevant, which process specifications are required? The basic requirements on data specification together form the *format* for the process data. Many different types of processes are involved in the life cycle analysis, ranging from raw material extraction, energy transformation, transport, forming, connecting, retailing, use and several types of waste management, including the transformation into final waste. Would different formats for different types of processes be necessary? This does not seem the case, since for all processes involved the indication of only three main functions appears to be necessary:

◇ the different types of environmental interferences of the process
◇ the different process products these are to be allocated to
◇ the further processes required for the functioning of this process.

Thus in these terms only a single format seems required for all types of processes, both in production and consumption and waste handling and recycling, that covers the three functions of the process data in the inventory analysis. The different treatment of such process types as energy production and transport, proposed by Vigon et al. (1993) seems to add only complexity, without systemic reasons. The very real practical problems, e.g. how to include transport, can be handled in practical procedures, not in the basic theory. Thus, only one format for all economic processes in LCA is necessary.

[1] This section is a revised version of part of Huppes 1992.

Assumed here are several general notions on processes, such as (sub-)systems that process inputs into outputs. In that case, only two basic distinctions are required if the format is to have the three functions specified. The first basic distinction is that between environmental inputs and outputs and economic inputs and outputs. Economic inputs are defined as inputs from other economic processes and economic outputs are defined as outputs to other economic processes. Environmental inputs are inputs to the process from the environment and environmental outputs run from the process directly to the environment. The boundary between the economic processes of a product as a group, i.e. the product system, and all related environmental processes, i.e. the environment system, have to be defined independently. I here suppose that such a definition exists, although it is certainly not selfevident[1].

The second basic distinction in process data in LCA is that between inputs and outputs that are to be allocated, and inputs and outputs that are to be allocated to. Environmental inputs and outputs are always allocated to certain economic inputs and outputs[2]. Thus there are two types of economic inputs and two types of economic outputs, those to be allocated and those to be allocated to. What is their distinguishing characteristic? It is a symbolic economic one: if taking an input from others or supplying the output to others is valued positively by these others, be they producers, consumers or waste managers, then all other inputs and outputs are to be allocated to these positively valued inputs and outputs. That is equivalent to saying that the process has a function for these others. In the practice of a market economy, this function or value is mirrored in a positive market price for the process considered, i.e. in its proceeds. To put it in still other terms, the *raison d'être* of an economic process lies in the proceeds it supplies for its owners. These proceeds cause the process, in a social sense, being a necessary condition for its existence. If there are no proceeds, the process will quickly come to a halt. All other inputs and outputs, both environmental and economic, are caused indirectly by the technological necessities of supplying the functions that realise these proceeds. All functions except that of final consumption may be expressed in terms of proceeds[3]. That can only be expressed in terms of itself, as the functional unit of the product, i.e. as the output of the use process of the product. All other processes required are attributed to these functions in LCA.

The main outline of the format can be represented graphically, as a template, see figure 4.2.2a and b. The process is a black box, with a name. The basic template defined here differs from that generally used in the USA, for example, by Fava et al. (1991, p.13), by Hunt (1992) and by Vigon et al. (1993, p.55), see figure 4.2.2c. There, no systematic

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[1] There are several problems associated with the boundary to the environment. Is wood production an economic process or does wood grow naturally in the environment assisted by human guidance? Has waste on a well-guarded dump site been emitted to the environment? Or is guarding waste on a site an economic process, with certain emissions resulting from leakage, evaporation and erosion?

[2] This is not strictly necessary. Governments might be involved in producing clean water, as is the case in the Netherlands where the water flowing into some nature reserves is first dephosphatised. In that case the one process function, described at the physico-chemical level, is taking some nutrifying substance from the environment. All other environmental inputs and outputs and all economic inputs and outputs would be attributed to this one function.

[3] This exceptional status of consumption in a material analysis is reflected in the statement of Fisher (1906): "The only true method, in our view, is to regard uniformly as income the service of a dwelling to its owner (shelter or monetary rental) the service of a piano (music), and the service of food (nurishment) ..." (emphasis in original, as cited by Kneese et al. 1970, n.8). The service is the function that the material product gives.

distinction is made among the environmental or economic types of processes where the inputs may come from and where the outputs may go to. Often, the origin and destination will be quite self-evident, as with emissions to air. In other cases confusion may easily result (and has resulted, in our own case studies as well), as with "raw materials" and "energy" inputs and "solid waste" and "waterborne wastes" outputs. These may originate from, and be supplied to other economic processes or from and to the environment. A second main point of difference is the inclusion of transport in the template of a non-transport process. In the general approach advocated here transport is a process like any other, requiring all the types of inputs and outputs specified.

Now, for current purposes, where the focus is on multi-output processes and waste processing, the description of a process in terms of six basic categories is still somewhat too general; they can be filled in with more mundane categories. First, products are economic outputs. Services are economic outputs as well. Examples here are transport, repairs, maintenance, and entertainment. Products thus cover both goods and services, as is usual in economics. Secondly, how does waste[1] fit into the template? Three types of waste may be distinguished in terms of the primary categories of the template, depending on its properties. Waste as solid waste may enter the environment directly as an environmental output. That type of waste is "final waste to soil". An alternative preferred term is "emission to soil".

A second type of waste may be an output of process x supplied to another economic process for free or for a charge to the purveyor, as a "product" with a negative value. This is a waste with costs for process x, or waste-to-be-processed by *another* economic process, whose processing service somehow must be paid for, through the market or collectively. Process x, the general process, may be that other process itself, so waste-to-be-processed is also an input category. This is the central meaning of waste here.

Thirdly, it may be a product that is sold at a positive price to another process, as any other goods or service, but with a relatively low value per kilogramme. An example is fodder produced in the medicine industry. This is "waste with a positive value". This type of waste output may also occur as an input, in the other process. It is very confusing to use the term waste in this sense, as the same flows may also be indicated as "product" or "co-product". The general term 'product' then is to be preferred. See figure 4.2.3 for the resulting refinements in the format and the corresponding template. Similar refinements in environmental inputs and outputs have been added. They have no specific function in the argument here. Since by definition, the economic *out*put of one process is the economic *in*put of another process, the economic output types are by necessity the same as the economic input types. Environmental inputs and outputs do not have the same symmetry[2].

---

[1] 'Waste' is often used in an ambiguous way. Most broadly, it may denote anything that people do not want and wish to get rid of. Such waste may be emitted to the environment, ranging from CFCs and carbon dioxide to sewer effluent and discarded refrigerators, and it may be delivered to others, for further processing and possible use. Thus waste is a very general term, covering all outputs to the environment and all outputs to other economic processes that are not seen as products. More restrictively, "waste" may also denote solid (or contained fluid) waste, as something that is supplied to other economic processes for further processing, i.e. intermediate waste, or that may be supplied to the environment as final waste. The second, more restrictive definition does *not* include emissions to water and air. Even so, the term remains too broad for unqualified use in the format.

[2] Sometimes they may, as is the case of phosphates emitted to surface water that are extracted back from surface water in drinking water production.

FIGURE 4.2.2 TEMPLATES OF ECONOMIC PROCESSES IN LCA

FIGURE 4.2.2A TEMPLATE OF AN ECONOMIC PROCESS IN LCA, WITH DISTINCTIONS RELATED TO ORIGIN AND DESTINATION (1) AND TO PROCEEDS OR COSTS TO PROCESS X (2)*

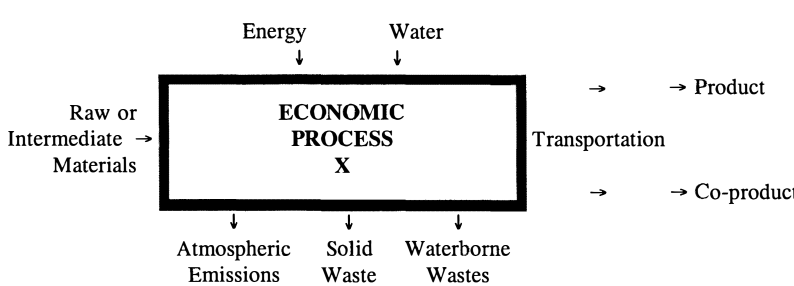
distinction (2)	*distinction (1)*		*distinction (1)*	*distinction (2)*
economic inputs, with proceeds, **to be allocated** <u>**to**</u>	inputs from other economic processes = **economic inputs**	**ECONOMIC PROCESS X**	outputs to other economic processes = **economic outputs**	economic outputs, with proceeds, **to be allocated** <u>**to**</u>
economic inputs, with costs, **to be allocated**				economic outputs, with costs, **to be allocated**
environmental inputs **to be allocated**	inputs from the environment = **environmental inputs**		outputs to the environment = **environmental outputs**	environmental outputs **to be allocated**

* The costs of process X are the proceeds of process Y, and *vice versa*.

FIGURE 4.2.2B FULL TEMPLATE, SIX MAIN CATEGORIES

economic inputs with proceeds to process x	**ECONOMIC PROCESS X**	**economic outputs with proceeds to process x**
economic inputs with costs to process x		**economic outputs with costs to process x**
environmental inputs		**environmental outputs**

FIGURE 4.2.2C BASIC TEMPLATE ACCORDING TO FAVA ET AL. 1991, HUNT ET AL. 1992, AND VIGON ET AL. 1993



In the main structure of the model developed in Part Two, a distinction has been made into structural, cultural and economic processes in society. The latter were split analytically into symbolic economic processes and physical economic processes. The format specified defines part of an economic model. The question now is to which layer does this specification of processes as indicated by the format belong, the physical, the symbolic, or both. First, one might assume that all inputs and outputs are specified in physical terms only. For environmental inputs and outputs this is certainly the case. For economic inputs and outputs the situation is not so clear. Descriptions may differ in this respect. A certain farm produce may variously be described as cow's milk, indicating its origin in the process; as water with sugars, proteins, etc., indicating its physical nature; as a refreshing healthy drink, indicating some of its functions; or as $0.80, indicating its sales value. The latter two aspects belong to the domain of the symbolic economy. The chemical composition defines the physical economic aspect. The origin might belong to either, as a systemic category, linking processes.

A theoretically thorough and thus practical descriptive approach has been developed for building materials. In an Aristotelian vein, the Swedish set up back in the Fifties the SfB system for product description, now used extensively in some form throughout Western Europe and Canada. In that system a format for products is specified according to its form (a 1x20x20cm tile), its material composition (fired, glazed clay), and its function (covering kitchen floors), each property to be detailed at some level. An architect's specifications might be restricted in the first stages of design to the function. But before a builder can order the tiles/fired clay/kitchen floor, the product has to be specified in all three respects, in much greater detail. Finally, a retailer may be specified, as the last-step originator, and the product's price. The destination is the kitchen in the housing project.

Some of these aspects can be used in the format for LCA. The form/material aspect does not readily indicate the source of the product or its destination in another process. The function gives the link between processes. The value plays a major role in allocation, by whatever method. No practitioner of LCA would allocate the emissions of a process to economic outputs with negative value, i.e. to wastes to be processed. In that sense all current LCA methods not only consider the functions of products but also their values, at least in a dichotomous manner. A minimum requirement for the description of economic inputs and outputs is that they cover physical aspects, such as form and chemical composition, and their function, by at least indicating origin or destination in other processes. "Low sulphur fuel oil" and "low fat consumption milk" are examples where chemical composition, origin and destination are effectively mixed. The minimal value aspect is already part of the format in the distinction between outputs with a positive and a negative value. The physical, the functional and the value aspect need not always be fully specified. For LCA, however, all three are required. Thus, LCA cannot be merely a physical analysis. Hence for LCA the knowledge format could be defined further in that the following elements be specified, each at some level of specificity, *for each individual economic input and output*:

1.	Name
2.	Function / value / origin / destination
3.	Form
4.	Material (chemical-physical) composition or characteristics

Specifying all categories will usually involve a certain element of redundancy. A product name may already indicate its function, origin, destination or form. Thus "kerosine" indicates an origin, and is limited to one form, fluid; it has a certain (but variable) chemical-physical composition, and a limited number of destination processes. Alternatively, "diesel oil" (now partly made from vegetable oils), does not indicate an origin but specifies the function and destination much further, such as an internal combustion engine of the Diesel type. It is only available as a fluid. Its material composition is variable. Some types of low sulphur Diesel may have exactly the same composition as kerosine. The most specific description is of a specific product as one input or output, of one real process, existing somewhere, at a certain time. Often, the name and specification of function and value will also indicate origin, destination and form with sufficient precision.

Some authors assume that the life cycle analysis should be restricted to the physical level, see Fava et al. (1991) and Vigon (1993) on allocation (i.e. level four in their inputs/outputs description). Strict adherence to this prescription seems impossible. How is a car to be described in its purely chemical-physical-spatial aspects? Or a fork? Or a telephone call? How can a product system be specified if not through the indication of its function in terms of the functional unit? It is the economic function as forming the link between the physical domain and the symbolic domain, that is required for policy instruments. That link can be made only by specifying aspects of both the physical and the non-physical, symbolic level.

This knowledge format covers the main distinctions required for the quantified computations in LCA. It does not yet cover the specifications defined in the goal definitions that ultimately are the basis for the validity of the results. Each product system is to be specified as to place and time, and according to further characteristics related to the group of products the system belongs to. Thus, all processes forming the product system should also be specified as to place and time, and as to the group of processes they represent. With these additions, the format is not yet specific enough for practical application. If data are to be verified, their sources should be available. This is a tedious procedure since the data for one process may come from many different, often unspecified sources. The broader function of these "data on data" is to help assess the validity and reliability of final results. No specified method for this validity analysis is available[1]. Only indirect checks can now be used, as through mass balancing, per process, and the sensitivity analysis described above. See Heijungs et al. for all further specifications of the format related to this validity and reliability. For the formal analysis

[1] The Wintergreen workshop on data quality organized be SETAC in October 1992 (no report is available yet) addresses the subject mainly in procedural terms and in terms of many single items on data quality.

of allocation now following, these further specifications are not relevant. The analysis will build on the format elements specified in figure 4.2.3, with economic inputs and outputs specified in both their physical and functional/value aspects.

FIGURE 4.2.3 MAIN CATEGORIES OF FULL TEMPLATE[1]

*intermediate wastes, with proceeds for process x	economic inputs with positive function in *other* processes		economic outputs with positive function in *other* processes	*goods (incl. secondary energy, materials, and intermediate wastes) *services with proceeds for process x
*goods (incl. secondary energy, materials and intermediate wastes) *services with costs for process x	economic inputs required for process x	**ECONOMIC PROCESS X**	economic outputs caused by process x	*intermediate wastes with costs for process x
*resources: -energy carriers (fossil/fissile) -other non-renewable -renewable *space *other environmental interferences	environmental inputs required for process x		environmental outputs caused by process x	*emissions to water and air *final wastes = emissions to soil *electr. magn. radiation *sound *other environmental interferences

4.2.4 Inventory: Allocation[2]

1. The allocation problem

Chlorine for PVC can be produced from brine only together with sodium and hydrogen. Which part of the energy requirements and other inputs is caused by the chlorine part of the production? The emissions of a household waste incinerator are produced by all wastes together. Which part of the emissions is caused by e.g., the carton/PE milk packaging in the waste? Recycling of waste, e.g. plastic bags, has waste disposal as a first function. As a second function that process produces a product, e.g. PE granulate, to be used elsewhere in another process for another product. Which part of emissions of the recycling process is to be attributed to the functioning of the product investigated and which part to the other product? In this section, an attempt will be made to solve the problem of allocation of these different types of multiple processes in one general theory, building on economic theories of cost accounting. Much work remains to be done before the result can become an operational method.

[1] All material flows go from left to right. Money flows towards the process from both sides, in the shaded rectangles, indicating the total proceeds or gross value of the process. Money flows out to both sides, through the other economic inputs and outputs, indicating the total costs of the process, apart from environmental taxes.

[2] This section is a much revised version of Huppes 1992. Since this section is a longer one, the subheadings introduced have been numbered.

The theory of allocation in LCA has not yet been extensively developed, although backed up by decades of practice. Vigon et al. (1993) maintain that the allocation should be on physical grounds, e.g. according to the share of the product in the total mass of all products. When results seem "strange" another measure can be used, such as the number of electrons taken up in electrolysis by the different resulting products (p.48). What are "strange" situations? Here are a few. The different products of many pharmaceutical industries, are typically pharmaceutical products, forming the main value of the process, and food and fodder, forming over ninety-nine percent of the mass. To which of these products are the sometimes nasty emissions to be attributed[1]? Allocation by mass would lead to the strange result that the pharmaceutical products hardly cause emissions. As another example of multiple processes is waste incineration. Incinerators for household waste emit heavy metals such as cadmium, mercury and lead. Over half the mass burned is kitchen waste, free of heavy metals. Assume again that mass is the basis for allocation. Should over fifty percent of the cadmium, mercury and lead emissions from the incinerator really be attributed to food? Or is this a "strange" result? Should the cadmium battery thrown into household waste, a tiny fraction by weight, be assumed not to lead to cadmium emissions? This can hardly be reasonable. Another type of multiple process is waste recycling. Recycling of copper by burning old wires, for example, is well known as an under some conditions extremely dirty activity. Wires go into the process, with some heavy oil. Several hazardous emissions, such as dioxins, and copper, the only product, come out of the process. The dioxins go into the air and the copper is used in alloys for kitchen sinks, for example. Mass cannot even be the basis for allocation here. To which of these functions of the process are the dioxins to be attributed, to the waste processing of discarded electric appliances or to the kitchen sink? Finding an answer to such questions is the aim of this section.

What are the restrictions to choosing an allocation method, which requirements are to be met? Easy solutions are not allowable. The inputs such combined processes require from other processes, the wastes they deliver to other processes, and the environmental interferences they cause cannot all be attributed to the one product analysed in an LCA, nor to the other product, not analysed. Nor can such processes be left out of the analysis altogether. A more reasonable but more complicated requirement for an allocation method is that, first, any unit of an environmental interference of a multiple process is attributed to one of its valued outputs as goods or services only, and secondly, that each unit of all interferences of a process is attributed to one of these goods or services. This second restriction is more related to the aim of the analysis.

The aim of the allocation analysis is to indicate which part of all economic inputs, wastes and environmental interferences is caused by the product analyzed. That causal analysis should answer the question what will happen if the product is made differently or is not made at all. If a correct causal analysis is made, there should be a reasonable expectation that the change from the one product system to the other will lead to the environmental changes indicated. The theory of allocation, though of a general and abstract nature, still has to be as faithful to truth in its empirical references as possible. A causal model of the whole economy would do the job, incorporating material and financial economic variables

[1] Allocation and attribution are synonyms here.

and also cultural and structural ones. Such a model in not available and will not become available for a long time to come. Also, it might not be practical to lump so many variables together. So, analytic criteria at a more micro level are to be designed that catch the main elements of this causation and are relevant for the questions asked in LCA. It seems advisable, first, to exclude cultural and structural variables from the analysis. These variables then might still play an independent role at a case level. Secondly, it seems advisable to restrict the symbolic economic part of the conceptual model to simple quantitative adjustments within the given technology, to assume linearity for all quantitative changes considered, and to make the model a comparative-static one, leaving out all dynamical aspects in time.

We now return to the central question of this section, *how are all inputs and outputs of any multiple process to be allocated to its different valued goods and services, i.e. to the different* ***external*** *functions of process.* Answering this question amounts to disentangling the causes and effects of the ways processes function. The main complicating factor remaining in this analysis is that part of the causation is symbolically economic, and another part is materially economic, that is physical. The *raison d'être* of any economic process is the value it produces. Without such a value it will soon cease to exist[1]. The shaded input types and output types in figure 4.2.3 indicate the functions that together create the total value a process may produce. All other economic and environmental inputs and outputs are thus the result of this positive value, are caused by it in a social (i.e. symbolic economic) sense. It has to be attributed to these shaded input types and output types. In a purely physical sense, however, all inputs cause all outputs, especially if the installation itself is taken as one of the inputs. This web of related symbolic economic causal relations and physical, economic-technical causal relations has to be disentangled so that each remaining input and output is caused socially, or physically, or both, by one of the valued products of the process. Any multiple process is thus dissected into a number of *constructed single processes*. Starting at that result, the basic restriction on the choice of allocation may now be reformulated: *the sum total of summing up all constructed single processes of a multiple process should exactly equal the full multiple process itself.* The causal analysis required here may also be stated in terms of a marginal analysis: what change in the emissions for example, of the one multiple process to be allocated would ultimately result if the functional unit were increased or decreased by a tiny fraction[2]. The independent variables are economically defined as the inputs and outputs that create proceeds for process x; all other inputs and outputs are the dependent variables caused.

The analysis now proceeds, first by specifying the allocation problem in more detail, in terms of the knowledge format developed, that is the template. Three basic types of multiple process result. The available approaches for allocation used in LCA are then surveyed. Next, a detour is made into cost accounting. In cost accounting the questions asked by managers of firms are very similar to those in LCA. In combined processes, the question is how all cost items of these multiple processes are to be attributed to the

[1] The value created may be a collective, in which case some form of collective payment may indicate that value.

[2] Some unclarity remains, as the marginal analysis itself is variable one, depending on the time horizon taken into account. The short term marginal effects includes only the variable elements. The long term marginal analysis includes all fixed elements. The latter seems the most relevant choice.

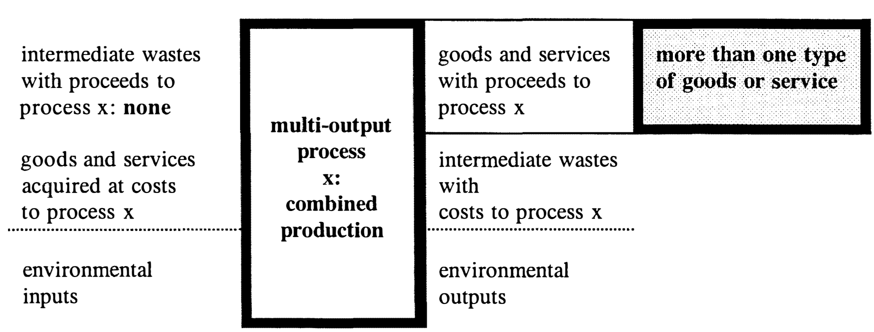
different goods and services produced. There seems to be a convergence in cost accounting to one method now, after centuries (!) of discussion, based on the share of each product in the total proceeds. That method is the *gross sales value method*. The new point in this cost allocation method is that it encompasses a methodological procedure. In several steps, a multiple process is broken down into subprocesses each of which requires specific treatment. Based on that method, which has been "normalized" here, the method for allocation, in LCA as well, is then formulated. Finally, these allocation principles are applied to the three basic types of multiple processes in LCA, resulting in the complete method of allocation. This long section concludes with a summary of the results.

2. The allocation problem detailed: three pure types

In which types of economic processes does the allocation problem occur? It seems quite widespread. Three pure types will be distinguished. It is based on the fact that a process may have valued inputs and valued outputs. Thus, there multi-output processes (combined production), multi-input processes (combined waste processing), and a third type combining one valued input and one valued output (waste recycling). Combined these pure types cover any multiple process empirically occurring.

The first pure type is that of a process with several products contributing to its total proceeds. Basic industrial production processes, which produce materials, very seldom produce a single type of output. Steel is produced in many varieties in the same installation. Oil refining and basic chemical industry are linked by an extremely large number of chains. More complex products like parts and components may sometimes be produced more easily in specialized installations, dedicated to a single product[1]. Usually, however, an installation is not dedicated to the production of only one product and supplies more than product. This combined production may be in the form of alternate products, like a welding machine applied to different tasks at different times. More complicated are the examples mentioned, where several products are turned out at the same time. Thus, the first type of process where the allocation problem occurs is that with several products made by one economic process: the *multi-output process*, see figure 4.2.4.

FIGURE 4.2.4 MULTI-OUTPUT PROCESSES: COMBINED PRODUCTION

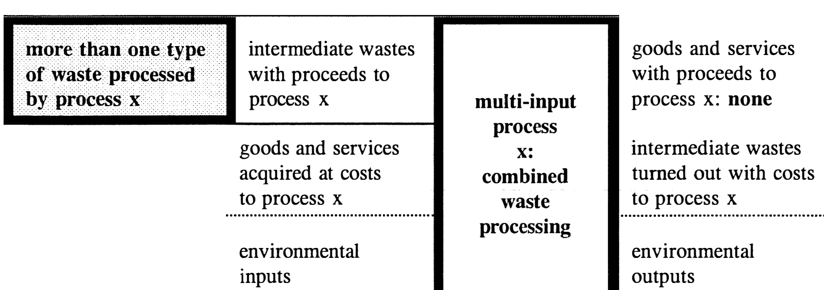


[1] In economic terms products that are equal prima facie may still be different. In the semi-conductor industry an extremely important distinction is that between the fully tested, 'all good' category and other categories with a certain or even uncertain failure percentage.

For this multi-output situation other terms are used as well, such as co-production. That name suggests that there is a main product and a co-product in the sense of a by-product[1]. That distinction is not relevant, however, in LCA allocation or in cost allocation as is widely accepted (Bierman et al. 1989, p.536, Heijungs et al. 1992b and Vigon et al. 1993, p.96). It is also not always applicable. In the example of the refinery there is simply no main product. The term "combined production" is used in the literature on cost allocation and will be used here for any process that turns out more than one type of good or service.

The second pure type is less common but still occurs many times in every product system. It is the processing of several wastes in the same process. The peculiar characteristic of pure waste processing is that its value lies not in the production of products but in their annihilation. Ideally, waste goes in and nothing comes out. Pure waste processing creates its value by processing *inputs* that others have no useful application for. Outputs, such as emissions and remaining wastes to be processed elsewhere, and other inputs, such as energy and materials required for the functioning of the process, have to be attributed to the different wastes processed. Only rarely will a waste processing installation be dedicated to the processing of one type of waste. It thus receives different types of wastes as inputs-to-be-processed, usually from several product systems: it is a *multi-input process*, for the combined waste processing of several types of waste, see figure 4.2.5. It is clear that at the case level the exact definition of what constitutes "a type of waste" is highly important. Household waste, for example, covers a wide variety of discarded products and other types of waste that may be defined in broader or narrower categories.

FIGURE 4.2.5 MULTI-INPUT PROCESSES: COMBINED WASTE PROCESSING

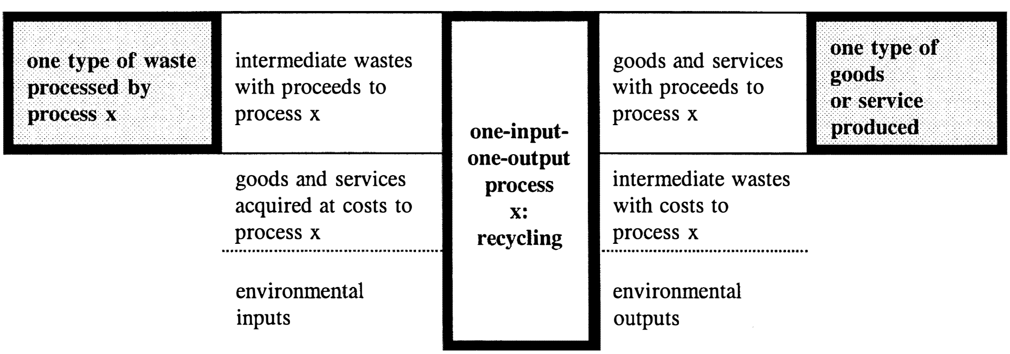


The third and last pure type of allocation problem is that in which a waste with positive value for the process receiving it is transformed into a usable product, through recycling or reuse. *One-input-one-output recycling processes* (covering both open loop recycling and closed loop recycling) are the final type of allocation problem, see figure 4.2.6. In recycling, the transformation may be limited, as in copper wire burning. Sometimes the transformation is more fundamental, as with sewer sludge processed into pig fodder and

[1] Vigon et al. 1993 p.97 define co-products as by-products. Heijungs et al. 1992b define co-products as the result of combined production, not as by-products.

ultimately into meat and hides. If reuse takes place in the product system analyzed, as for example with reusable milk bottles, the loop is closed. If there are no other multiple processes involved in the loop, it does not make any difference for total results which proportion of the impacts of the process is allocated to the one or to the other process in that product system. This, however, is an exceptional case. If other multiple processes occur in the loop, this simplification is not possible. The method of allocation for the recycling part of the process then does influence total results, i.e the distribution of environmental interferences between product systems. If negatively valued waste is processed into products used in other product systems only, the loop is *open*. In this more common situation the choice of the method for allocation is of course essential. Often the term recycling also refers to products produced from "waste with a positive value". In the format developed here, a waste with a positive value is not a category to be used. Such an output is not a waste at all but is one product from a multi-output combined production process and for reasons of allocation belongs to the first pure type.

FIGURE 4.2.6 ONE-INPUT-ONE-OUTPUT PROCESSES: RECYCLING



In the systems approach developed here there are no other types of multiple process, only combinations of the types described. Three such combinations may be formed, always involving the last type, the one-input-one-output type of recycling as the connecting process type. First, it may recycle an intermediate waste into several goods or services, as with steel and non-ferrous metals from car scrapping. That is a combination of recycling and combined production. Or it may use several intermediate wastes as input, while turning out only one product, as with methane production from household sewer sludge. That is a combination of recycling and combined waste processing. Or it may produce several products from several intermediate wastes, as with the agricultural production of milk, hides and meat, as products, with sludge from waste water purification installations and the offal from potato chip manufacturing as two negative valued inputs. That is a combination of recycling with both combined production and combined waste processing.

3. Available methods for allocation and their limitations

Which methods for allocation are now available for the three situations distinguished, and why should another be developed? Allocation in *multi-output processes* has received much attention. The properties explicitly named to define the ratio of one product's share in the total impacts of a process are:

◇ mass (most studies)
◇ number of molecules (e.g. Fraunhofer Institute and Battelle/Franklin)
◇ inherent energy (e.g Kindler and Nikles 1980)
◇ number of electrons (Vigon et al. 1993)
◇ value (Basler & Hoffman, in 1974 already)
◇ function (e.g. Frischknecht et al. 1991).

Allocation has most generally been defined in purely physical terms, mostly by the ratio of mass of output. The choice of mass needs further specification, however, to become operational. One possibility is to take the ratio of the one product studied to the total mass of all outputs, including wastes (as suggested in a draft version of Vigon et al. 1993). The argument used is that in that way no reference to the economic qualities of the outputs is necessary, only to their physical aspects. Alternatively, the ratio to the total mass of all valued products may be used, excluding at least the mass of negatively valued outputs. The allocation problem is most pertinent in joint production where the whole process inherently produces several products. Sodium chloride cannot be split into sodium without chlorine resulting as well. About this example, Vigon et al. (1993, p.57 and p.58) state:

> *"There is no entirely satisfactory solution but common practice based on chemical engineering, chemistry, and physical experience is to apply a mass allocation scheme as a reasonable modelling technique".*

Generally, they suggest that when a more specific method, based on physical and chemical processes, is not available,

> *"... a simple mass allocation can be used".*

Why could such "simple" or "traditional" methods for allocation of multi-output processes, such as those based on mass, number of molecules, or inherent energy not be used primarily? There are several arguments against the general use of these physical units in allocation. First, a choice between the physical types is quite arbitrary and may lead to very different results. Where several methods of allocation are applicable, as is true for example in refineries and in the joint production of caustic soda, chlorine and hydrogen, the outcomes may differ sharply between methods. See figure 4.2.7, in which different methods of physical allocation have been specified for a multi-output case, that of chlorine, sodium hydroxide (caustic soda) and hydrogen production. For comparison, a value-based method of allocation has been added. The share in emissions, energy use, etc., allocated to chlorine, one of the products, may differ by a factor of 3, depending on the physical allocation method used. Such a margin seems unacceptable for an allocation method.

Secondly, different positively valued outputs of a certain process may be expressed in terms of different physical measures, e.g., mass and energy. In such a case it is impossible to compute proportions. It is common for a process to produce both a material product, with a mass, and water and steam, with a low or medium caloric heat. Example are glass and glass products, and steel and steel products. These material products may have little or no inherent energy, while the co-produced energy need not have any mass to speak of (heat and electricity). How would hot water from one process, used in a secondary process, and then returned to the first process with a lower temperature be treated, by mass or by energy?

Thirdly, in many circumstances physical allocation based on mass, moles or energy is not possible at all. How could one allocate transport, welding, storage, etc.? Services, as a major sector in the economy causing environmental interferences, produce goods of a non-material nature such as R&D-based technical advice, telephone calls and transport. How are car leasing and car rental to be handled? Introducing additional physical properties may solve the problem in each case. However, the number of physical measures would then also increase and the next choice is more difficult, leading to more problems of the first type. There is already non-applicability with respect to any simple physical measure if one of these outputs is a non-material service.

Fourthly, where applicable, the results are often strange, as in the case of the emissions of the pharmaceutical company that were allocated to the co-produced fodder.

Thus the different physical methods of allocation have the following problems:

◇ different methods that are applicable may lead to very different results in the same case
◇ different methods are applicable to different products but no single method is applicable to all
◇ physical-based methods often may not be applicable at all
◇ in many cases results are intuitively unacceptable.

These problems in allocation in multi-output processes seem quite difficult to solve in the purely physical approach. At least a higher level criterion should specify which sub-method to apply and when.

The non-physical alternatives named are the value-based method and the function-based method. The function approach has often been used implicitly, disguised as a physical approach. The distinction between the purely physical and the functional approach is that between a purely chemical-physical description in terms of SI (Système Internationale) variables, and one in terms of functions supplied. The difference may be quite subtle. *The square meters of storage room* required in retailing may be the basis for allocating the share of a product in the shop. It may be looked upon by some as a physical measure, as the SI variable *metre* is used. The dimension, however, is *storage room*, i.e. a functional description of what the warehouse does for the product investigated. In fact, all services that have been treated in LCA have always been allocated on the basis of a unit of function[1]. Diverse examples are transport, welding, washing, heating, and data communication. The use of that measure, however, has been limited to single output processes, since by definition the function of one output can never be the same as that of the other.

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[1] The *functional unit of product*, the basis for quantification and comparison in LCA, is exactly such a function. The problems of selecting one function for comparison quite arbitrarily from among the several functions common to most products, similarly occurs in allocation. It occurs in real life as well in the specification of the market price of the product related to some characteristic.

The main problem with methods of allocation based on physical causation is that, contrary to motive related social causation, it cannot work backwards in time. Thus an output can never cause an input physically. If some product of a multiple process is not sold but delivered to a waste processor instead, *there is no physical change in the process studied whatsoever and purely physical-based methods could not register a change*. The allocation has to involve the social, i.e. economic, aspects of the process, as non-physical entities. In allocation-based reasoning in terms of the physical sciences, it only is possible to allocate outputs based on inputs, since physical causation only goes in the same direction

FIGURE 4.2.7 DIFFERENCES IN ALLOCATION TO CHLORINE IN THE JOINT PRODUCTION OF CAUSTIC SODA, CHLORINE, AND HYDROGEN DUE TO THE METHOD USED

Process (outline)	*Method of allocation*[1]	*Resulting shares in allocation*	*Index share of Cl, with method (2) = 1*
2 NaCl + x H2O + Energy ↓ 2 NaOH + Cl_2 + H_2 + (x-2) H_2O	**1. Mass of atoms from the split-up salt molecule** Na (23) Cl (35.5)	[Na $\frac{23}{23+35,5}$.100] = Na 39.5% Cl 60.5%	[60,5% : 20%] = **3.03**
	2. Number of atoms: Na (2) Cl (2) H (4) O (2)	Na 20% Cl 20% H 40% O 20%	[20% : 20%] = **1.00**
	3. Mass of molecules: 2(NaOH) (80) Cl_2 (71) H_2 (2)	Na 52% Cl 46% H 2%	[46% : 20%] = **2.30**
	4. Mass, including water:[2] NaOH.4H2O (112) Cl (35.5) H (1)	NaOH 75.5% Cl_2 24% H_2 0.5%	[24% : 20%] = **1.20**
	5. Number of molecules (excluding water): NaOH (2) Cl_2 (1) H_2 (1)	Na 50% Cl 25% H 25%	[25% : 20%] = **1.25**
	6. Mass including packaging, inherent energy, electrons, other ? ? (?)	??	
	7. Gross sales value:[3] NaOH 45% Cl 50% H 5%	NaOH 45% Cl 50% H 5%	[50% : 20%] = **2.50**

[1] I could not reproduce the electron-based method mentioned of Vigon et al. 1993.

[2] The amount of water has been arbitrarily chosen so that the percentage allocated to chlorine nearly coincides with the figures in Brown et al. 1985.

[3] These relative values reasonably represent long-term shares in proceeds according to personal communication with AKZO, the main producer of NaCl in the Netherlands.

as time. Inputs always precede outputs in time since they are transformed into outputs in the process. So, by the logic of the causation involved in physical sciences, purely physical causation can never solve the allocation problem of combined production and recycling. It might play a role however in multiple waste processing.

Multiple waste processing has generally not yet been dealt with specifically. One exception is Hoefnagels et al. (1992). In this study, in which CML was a partner for software development, emissions from household waste incineration were allocated on the basis of mass. This leads to strange consequences. Imagine a household refuse incinerator burning 1000 kg of kitchen waste, 10 kg of PVC packaging material, and 1 kg of discarded nickel-cadmium batteries, containing 0.5 kg of cadmium. The result would be, *inter alia,* emissions into the air of CO_2, dioxin, and cadmium. Allocation by mass would assign the CO_2 emissions, the cadmium emissions and the dioxin emissions almost exclusively to kitchen refuse. For CO_2 this is almost correct. For cadmium, a direct physical causation can easily be constructed that is extremely at variance with the allocation by mass. Dioxin, formed from the soot of kitchen waste and chlorine from kitchen waste and PVC, are a much tougher problem for allocation since even the physical causation is disputed. Here the social causation and the physical causation work in the same direction, making several different social and physical allocation rules applicable.

Allocation of *recycling processes* has been specified only rarely. In the SETAC-workshop on LCA in Vermont in 1990 the very reasonable advice was to split the direct impacts of recycling on a fifty-fifty basis between the waste producing product system studied and all other product systems that use the recycled products. This practical advice was given with a proviso, as being valid as long as no better method becomes available, see Fava et al. (1991). The general analysis of the recycling problem has usually been framed in a most unfortunate manner see Fava et al. (1992) and Heintz (1992). The allocation problem in these studies has not been approached as a problem of allocation of one process, as has been the practice of all authors when allocating multi-output processes and multi-input processes, but implicitly or explicitly, at the level of comparing different product systems.

Two situations are then generally discerned, that of open-loop recycling and that of closed-loop recycling. In closed-loop recycling the output of the recycling process is applied in the same product system. It is assumed by all these authors that the method of allocation does not make a difference in the case of closed-loop recycling. That assumption is unjustified. It is not correct when there are other pure types of multiple processes involved, as will often be the case. Suppose a process delivers a product within the system analyzed, milk, a next product in another system, meat, and also produces a negatively valued waste, manure. The manure is recycled and then used in fodder production. Part of the waste, the manure, is attributed to the other product system, through some allocating mechanism, e.g. mass or value, for the allocation of manure to milk and meat respectively. Manure recycling is a closed-loop recycling process, involving large emissions of ammonia. The recycling process results in "fertilizer" used in the production of fodder, that in turn is partly eaten by the cow. The recycling loop is fully closed, but several combined products go to other systems. Indirectly, part of the emissions from recycling the negatively-valued waste, from manure processing, are thus attributed to the other product system as well, e.g., that of meat. Now imagine that the recycling allocation rule applicable to manure processing changes. More emissions are allocated to the waste producing milk/meat production process (= cow) and fewer to the recycling process that results in fertilizer for the fodder production process. The other product system, the meat one, will then receive a larger share in the allocated recycling emissions[1] and the milk one a correspondingly smaller share. Thus closed-loop recycling requires a non-arbitrary allocation rule as well, applied to any multiple process in the product system. More technically, not allocating in the case of closed-loop recycling makes any sensitivity analysis and marginal/improvement analysis at the level of individual processes impossible[2].

The current allocation in a single open loop recycling process has centred on a comparison of *four* different product systems:

◇ the product system investigated with the waste to be recycled in the recycling process
◇ the product system that uses the product of the recycling process
◇ the product system with the same waste, but hypothetically without the recycling process
◇ the hypothetical product system that does not use the output of the recycling process but uses primary sources of the material.

A comparison of the two actual situations together, including recycling, with the two hypothetical situations together, without recycling, gives a measure of the effects of *introducing* recycling. This involves an LCA of the two product systems together, one variant without the recycling process and one variant with the recycling process. It shows the environmental effects of introducing recycling. How true that analysis may be[3], it cannot give an answer to the allocation question since the two (and usually many) product systems involved are treated only together. Which part of the inputs and outputs of the recycling process should be attributed to the one and which to the other actual product system? Answering that question first requires the solution of the allocation problem at the level of the individual process! That allocation rule, apart from the provisional fifty-fifty rule, is not now available. This problem in the allocation method based on the comparison of hypothetical systems seems to have gone unnoticed. A number is produced, but not one based on allocation.

The conclusions here on the allocation options specified in the literature are that

◇ for certain situations several conflicting allocation rules are applicable
◇ in other situations only one rule is applicable, different for each case
◇ in some situations no allocation rule is available at all.

The value-based method, rarely expressed, may be applied in many situations to which others cannot, as worked out below. However, it cannot be the sole panacea. In multiple waste processing, payment for the processing of the different wastes, related to the share in costs, is the indication of their (negative) value. The resulting share in emissions is not reflected in that value. If cadmium batteries in household waste are the sole source of

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[1] To complicate this analysis part of the fodder is not for cows but for pigs, figuring in three other product systems.

[2] This is the case quite evidently with the mathematical part of that analysis, e.g. in the marginal analysis specified by Heijungs et al. 1992b.

[3] It is not, since the model mechanism is much too restricted. Introduction of recycling will lead to many adjustments in the processes 'upstream' and 'downstream'.

cadmium emissions of waste incineration it clearly is the cadmium batteries that cause the cadmium emissions. Any measure based on costs, or mass, would clearly lead to nonsensical results. In some instances allocation is based on the function produced, as with many services. I hope to show that both the function-based method, and the value-based method, and the physical methods, all may be subsumed under a single general method, thus solving the problems indicated. That general method thus would not only cover combined production but also multiple waste processing and recycling.

As an approach to generating solutions, a description follows in the next section of the diverse and complex allocation methods used by economists in cost accounting. Learning from similar experiences is a good method. Some parts of this section are quite technical. Readers not familiar with economic terms may wish to skip it and proceed directly to '6', allocation in the three pure types of multiple processes.

4. Methods for allocation in cost accounting

The main function of cost accounting[1], as part of operations research, is managerial support in decisions related to products[2]. The information generated from available data finds its application in guiding choices on processes, products and marketing. Managers need to know what effects on their costs and proceeds the decisions related to their product will cause. In that sense cost accounting has an aim similar to that of environmental LCA. Costs and proceeds cover all economic inputs and economic outputs. The subjects of cost accounting and LCA-inventory allocation also overlap. In cost accounting the problem of multiple processes goes under the heading of *joint products*.

Joint products are products coming from the same production process. A primary distinction can be drawn between joint products with joint costs, joint products with indirect costs, and joint products with separate costs[3]. With *joint costs* it is technically impossible to divide the process into subprocesses and produce each output in a separate installation. An example is the combined production of pig skin and pig meat. With *indirect costs* there is a technical possibility to specialize in each product separately. Combined production may be attractive, however, from an economic point of view. An example is the combined air transport of passengers and freight. All overhead costs are part of indirect costs. Overheads usually involve services of some kind such as repair, maintenance, marketing, and staff functions such as representation, documentation and research and development. And, finally, there may be *separate costs*, when each product requires its own specific cost items. In one firm, on one site, all cost types may be present, see figure 4.2.8.

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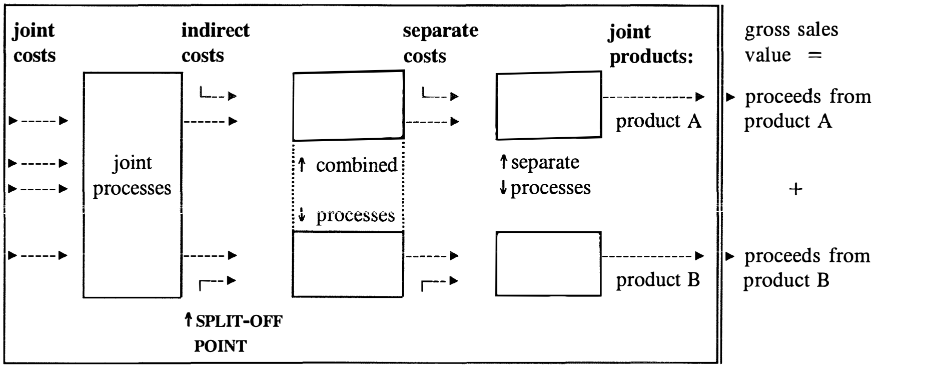
[1] I draw on an authoritative treatment of the subject by Bierman, Dyckman and Hilton 1990, especially chapters 14 and 15. Similar treatments may be found in any of the more extensive introductory books on cost accounting. See for example the relevant chapters in Drury 1992; Horngren and Foster 1991; and Raiborn et al. 1993, esp. pp.713-23.

[2] In environmental LCA, standardization is a separate problem, since interests in the method to be applied diverge. Everybody likes the systems that may make his products look good even if in fact environmental interferences are different. In cost accounting use of the wrong system leads to the wrong decisions in economic terms. In private industry this leads to relative losses. Such a corrective mechanism is not operative in LCA.

[3] Joint costs and indirect costs together are often called *common costs*.

FIGURE 4.2.8 JOINT COSTS, INDIRECT COSTS AND SEPARATE COSTS IN ONE FIRM WITH JOINT PRODUCTS



The fundamental issue behind economists' reasoning is that the *causation* of costs has to be determined. Causation states how a change in one variable leads to a change in another. This causation is dependent on the type of decision at hand, since these decide which mechanisms may be operative. In the airplane example, the decision on the size of the hardware, i.e. the fixed costs of the plane, is based on long-term (expected) use. On the other hand, decisions on how to operate the air service[1], involving the variable costs, are based on short-term use. To select a relevant allocation method the *time horizon* of the decision at hand needs to be determined. In LCA, it seems difficult, if at all possible, to differentiate the environmental assessment of a product in terms of the exact time of the processes concerned. Real short-term considerations are thus irrelevant. Is it long-term considerations that count in LCA only? Typical applications of LCA are those in product design, improvement analysis, marketing and eco-labelling. It is either totally different products that are compared or similar products that are varied in some respect. The relevant question might be how decisions work out "in the end", if one functional unit more of the product investigated is used. That is the same as asking what the long-term marginal effects will be. Of course the short term option is there. One could ask what the effect would be tomorrow (and next week, next month, etc.) if today I changed from product A, say milk cartons, to product B, say returnable glass bottles. However, nobody seriously proposes such a dynamic analysis. The conclusion here is that *allocation in LCA is about long-term marginal effects*. The distinctions economists make between short-term and long-term allocation may thus be further disregarded. Only long-term allocation is treated.

Joint costs, indirect costs, and separate costs may each be allocated differently. For the allocation of joint costs three possibilities are mentioned, one physical-functional and two economic. The *first* method is to allocate joint costs on the basis of some *physical unit* common to all outputs. Thus economists also contemplated the methods currently used in LCA. This physical approach does not usually relate to any of the aims of managers of

[1] Any service may be formulated as the output of a process that transforms one or more material inputs into the service. The use of a product is a service. The functional unit of product thus is a formal description of the service a product system provides for its user.

firms. Therefore it can hardly play a useful role in a decision support system. Only in special cases, when weight or energy is the characteristic on which the value is based, may the physical measure be an indicator of function and value. The sharp increase in the relative price of heavy bitumen, due to technological changes in further processing, has changed the status of this former waste product into an attractive refinery product. Purely physical measures, such as mass and inherent energy, cannot reflect this very real change in function and will remain unchanged. The physical measure thus cannot be generally adequate.

The *second* allocation principle is based on a measure of profitability, the *net realizable value*. The net realizable value per unit is the selling price less the costs of completion and sale, after the split-off point. The split-off point is the point in the production process where the joint products can be identified separately. From that point onwards decisions affecting the further processing and sale of the one product may be made independently of the other product in a technical sense. The net realizable value is one way to quantify the value of the outputs of the purely joint process part of a production unit. It does have a certain drawback, however. Suppose that the costs of further processing, after the split-off point, are not in the same proportion for all joint products. The apparent contributions of each product to profits then differ. The net realizable value method leads to changes in the joint costs assigned to one product if the separate costs, after split-off, change for the *other* product. This is because the net realizable value method assumes that the variable finishing costs of any product do not contribute to the fixed costs of the joint process and to profit. They generate only enough revenue to cover themselves. The method is nevertheless used in certain industries[1].

The *third* allocation principle for joint costs is also based on a measure of profitability, the *gross sales value*. The share of a product in total production costs is assumed equal to its share in total sales. It is assumed that all the costs incurred are equally effective in their

FIGURE 4.2.9 SITUATIONS FOR COST ALLOCATION DISTINGUISHED BY ECONOMISTS

problem type:	***subprocess characteristic:***	
separate costs	1	*technically separate processes
indirect costs	2a	*attributable combined processes (one function) -in-firm services
	2b	*non-attributable combined processes (several functions/non quantifiable) -operation -depreciation -maintenance -other non-attributable
	2c	*nearly joint overheads
joint costs	3	*pure joint processes

[1] Bierman e.a. indicate that in the meat packing, canning and mining industries, for example, the method is used in an even less acceptable form, where the physical aspects do not properly indicate function.

contribution to fixed costs and to profits. This holds for both joint costs, for combined costs after split-off and for separate costs for different products after split-off. The gross sales value method is first used to subtract the proper cost amounts of separate costs and attributable indirect costs (including their share in profits) from the respective proceeds. The remaining costs are the purely joint costs. These then may be allocated according to the share of each product in the proceeds of the joint process part as computed. The gross sales value method is quite simple to apply. The share to be allocated to one of the products is its share in the (computed) total proceeds.

For allocating the joint costs the gross sales value method is preferable to the net realizable value method. Physical units for allocation make no sense at all. Of course, the units to which the costs apply are to be expressed in empirical categories such as functions or physical entities, different for each of the two products.

For the allocation of indirect costs, four possibilities are discussed, three related to function and physical characteristics related to it, and one based on value.
A first allocation principle, based on a *functional characteristic of the cost item to be allocated*, is the *direct method*. The physical characteristic indicates the amount of the function of the item. All overhead input of a certain type may be allocated in the same proportion as the main production activities directly using that input. Suppose that product A requires 1000kWh of electricity and product B 1500kWh, separately, and that there is an overhead electricity use of 2000kWh. According to the direct method this overhead may then be assigned at 40% to product A (1000 /1000 + 1500) and at 60% to product B (1500 /1000 + 1500). Now what if the overhead input is of a type not used by the main production process? In the example this would be the case if production of products A and B did not require electricity. The direct method could not then be applied. The direct method is still used in some industries. Economists, however, no longer advocate it.

The second allocation of indirect costs, based on a *combination of functional characteristics*, is the *step-down method*. Again, physical characteristics indicate the amount of *function* of the items. Suppose that the overhead in the electricity example is for heating and lighting in the buildings of several service departments. Electricity used by each overhead department may now be assigned in proportion to its service to each production process. An example is the allocation of electricity use by the telephone department to product A and product B, based on the number of calls made by the separate sales departments for each product.

The third allocation of indirect costs, that considers *all inputs and outputs between indirect costs units*, is the *reciprocal services method*. It is a further refinement of the step-down method. Each overhead department responsible for indirect costs may supply its service to a production department, but also to another service department. In the same example, the electricity consumed by the maintenance department is in part for the maintenance of the telephone department. This indirect electricity use can be charged in proportion to the use of the telephone service. These parts of the process then are first separated from the total through the application of the gross sale value method. Thus made independent, their contributions to different other parts of the process are allocated according to the single function they have for each of the two products.

Finally, the allocation of indirect costs may be realized by the *gross sales value method.* The flying costs of an airplane, caused by passengers and freight together apart from the specific costs attributable to each, can be allocated to the shares of each in the remaining proceeds. Nonetheless, a measure is required for each as a unit to assign the costs to (e.g. ton-kilometre and passenger-kilometre). The gross sales value method is the only one that also can be applied to a special type of indirect costs, to overheads[1] that are *nearly joint.* Such overheads do not provide their services in concrete units to each product. Examples are the depreciation of fixed installations and the insurance payment for liability on total operations.

For the allocation of separate costs, the third type of costs, it is again the gross sales value method that lifts them out of the total process and allows their independent analysis. The gross sales value method was used in the same way to allow the application of the reciprocal services method. Here the contributions of the thus separated processes are not reciprocal, they relate to only one of the products. Their quantification is in terms of function, which is often made operational together with some physical measure, as is the case for any single process.

The possibilities treated by economists cannot all be relevant. If the most preferred option for each situation is chosen and the procedure to arrive at each situation is included, a generally applicable, normalized allocation principle for costs results. That procedure summarizes the economic ideal of cost allocation. It also specifies the ideal method for the allocation of economic inputs and outputs in environmental LCA.

The most complicated process for allocation is that in which all types of subprocesses thus far mentioned occur together. If the cost allocation procedure can handle that situation, it also can cover any simpler situation. The encompassing allocation procedure is to divide the total process into the subprocesses of each subtype, defined in relation to the allocation method used. Five steps may be distinguished in the procedure.

Step 1
Starting at the easy side, the right hand side in figure 4.2.8, separate costs are first split off. They concern those parts of the process that contribute their function only to the one product considered, fully independently from contributions to the other product. Thus the function is split into subfunctions that *are technically genuinely independent.* The subfunctions then can be treated as any separate process, with the quantification of the function based on the most appropriate functional units, that may or may not be expressed primarily in physical terms, e.g. "number of spoons formed" and "amount of stainless steel scrap molten", for the outputs of spoons from a spoon factory and its output of bars of recycled material respectively. If these were the only parts of the process, it would not be a multiple process at all, but two single processes. In that case, no allocation at all would be involved after separation. In the more complex situation the first step splits off single subprocesses where no allocation is involved anymore, and leaves a process part that is still combined and joint.

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[1] The terminology here is quite confusing. Economists see all overheads as the main elements of indirect costs, as opposed to joint costs. The overheads of the joint part of a process have the same characteristics as the joint process itself and might for that reason be better subsumed under joint costs. This will effectively be done hereafter.

Step 2
From the process body now left, another part is split off, the first type of indirect costs, part 2a. It concerns those parts of indirect costs that contribute to both products, but in attributable functional units, e.g. "total time of telephone calls". Technically, the contributions to each of the two products could be made independently, but they are combined for economic reasons of efficiency, e.g. one telephone operating system for all calls. Again, the subfunction specified may be expressed using physical terms, indicating the function at a technical level. This second process part thus also might concern a physical aspect of the subfunction, which is scarcely ever mass. Only if the function is "having mass" is mass the relevant aspect for the allocation operation, as with a filling material for a *presse papiers*. Allocation may be in some function terms, e.g. the number of telephone calls or their total duration, or in value terms, as the share of the costs (plus standard profits) of all telephone calls. The latter method is the more relevant. If the latter is unavailable, the functional specification in the most relevant terms is the best approximation.

Step 3
From the remaining process elements, the second part of indirect costs is split off, part 2b, that is those parts that cannot be attributed to both products in terms of the same function. The underlying physical relations might seem to support such a functional allocation, but only technically speaking. In the airline example, passengers might be measured in numbers or also by mass. Freight might be measured by mass, or also by number of pieces. However, these are not relevant units of function that effectively reduce both products to a relevant common denominator. This single process part has two functions, contributing to product A in one way and contributing to product B in another way. All inputs required are thus required for both functions, produced together. Such purely physical properties cannot be relevant since even the functional characteristic of "having mass" does not effectively apply to both products. Since the functions are to be defined differently, their relevant physical properties can only be different as well. Such different functions can be reduced to a common denominator only at a higher level of abstraction, i.e. that of the values of each of the functions produced. Their costs then can be divided into shares proportional to the two values, as proceeds. That is the only reasonable option available here.

Step 4
From the remaining process elements the "nearly joint" overheads are split off, part 2c, that is an overhead for which an attribution of its function to each product cannot be quantified at a functional level, let alone at a physical level. Examples are the costs of activities for making a brand name known, the costs of the secretariat of the Board of Directors, and the fire extinguishers for the joint installation. The function may be clear, its quantification is not possible other than on the basis of the gross sales value method.

Step 5
The remaining process element is fully joint, that is fully 'collective'. Even technically, it is not possible to split off parts that help create only one function[1]. All parts are entirely interdependent. This single subprocess produces two joint functions, one for product A and one for B. All the inputs required are thus again required for both functions, produced together. The two functions can be reduced to a common denominator only at the higher level of abstraction, at that of the value of the functions produced by the two functionally and physically different products. Again, their costs then can be divided into shares proportional to the two values, as proceeds, according to the gross sales value method.

See the survey on allocation of costs in figure 4.2.9, fully coinciding with the LCA allocation of economic inputs and outputs. It may be noted that economists only discuss the allocation of *costs*, i.e. of all economic inputs and outputs with a negative value for the process studied. The reasoning required for the allocation of environmental inputs and outputs might be different.

5. A general method for allocation in LCA
Any linear allocation method allocates part of all "other" inputs and outputs to each of the valued inputs or outputs by multiplying by a certain ratio. The ratio is the amount of some property of the goods or service studied to the total amount of that property in all positively valued goods and services of the process. In this sense even physical methods of allocation presuppose a value principle, to distinguish such goods and services from all other inputs and outputs produced by the process. "Purely" physical causation rarely makes sense in allocation; it does so only in some aspects of waste processing, see below. In a physical science sense all inputs, together with the transforming characteristics of the process, are always the efficient cause of all outputs. Physical causation cannot "go back in time" and thus can never play a role in the allocation of inputs to outputs. In the allocation of outputs to inputs, as in waste processing, physical causation might play a role. In a social sense, the value created causes the process. Its costs, i.e. the remaining economic inputs required and the unwanted outputs that cannot be disposed of in the environment, are caused socially by the value created, as are all its environmental inputs and outputs.

In allocation for LCA as well there are thus fundamental reasons against the principle of physical allocation, apart from the aforementioned practical considerations regarding its applicability. The main objection is that mass, moles, energy, etc., aspects of the goods produced, are not in themselves the objective of any process. It is only incidentally that they do reflect the real aims of the operators of economic processes. Their aim is to create a value for others, who use the product produced in their processes. Payments by others form the (symbolic) economic basis for the functioning of the process studied, forming its proceeds. If no value is created, the process will soon cease to exist and no environmental interferences will result. In a social sense all other inputs and outputs of a process are caused by the value of the products (goods and services) the process creates.

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[1] Of course one could throw away the meat of the one sheep and the hides of the other. Then, in a very artificial way, all joint costs would be reduced to the direct costs of the then single process.

However, that value is not an independent entity, it is based on the functions that the products of the process studied help to fulfil in other processes. Many functions have a material aspect in which they may be expressed equivalently. With bulk monomers this is quite feasible, as in "tons of substance X produced". Expressing such a materially complex product as a car in terms of purely physical units is next to impossible. Even with simple products such as a spoon, a purely physical description can hardly express its function, and certainly not its value. In multiple processes, physical properties only seldom indicate shares in the total value created. Allocation in LCA should reflect this state of affairs and cannot be purely physical.

The general principle of allocation, in both LCA and cost accounting, has been formulated as specifying those inputs and outputs that are caused by one valued input or output. For economic inputs and outputs, (products) a more specific formulation may now provisionally be given:
The function that the positively valued products of a process have in ***other*** *processes constitutes the basis for allocation.* That function forms the bridge between the physical-material aspects of the economic process studied and the value it creates[1] at the symbolic level.

FIGURE 4.2.10 PROPOSED METHOD OF ALLOCATION IN LCA, FOR SITUATIONS IDENTIFIED BY ECONOMISTS

process parts identified by economists:	***proposed method of allocation in LCA, for allocating economic inputs and outputs:***
separate costs:	
part 1, technically separate processes, required for one product only	**no allocation required*, function possibly made operational in terms of physical characteristics for reasons of quantification
indirect costs:	
part 2a attributable combined processes, with one function specifiable and quantifiable for both products contributed to	*function, made operational through *gross sales value method or function characteristics*; in very exceptional cases physical measures such as mass
part 2b non-attributable combined processes, with several incommensurate but quantifiable functions or one non-quantifiable function	*function, made operational through *gross sales value method* only
part 2c nearly or fully joint overheads with one function specifiable but no attribution possible	*function, made operational through *gross sales value method* only
joint costs:	
part 3 purely joint processes, one product physically necessary for the other	*function, made operational through *gross sales value method* only

[1] Alternatively, one could take the value as the central item and have a two tier process of becoming operational, first into functions and then, sometimes, into physical terms. Two tier processes requires awkward formulations and may easily lead to mistakes in reasoning. Therefore, the function has been chosen as the central parameter in allocation.

Cost allocation used by economists has added a dimension to the allocation analysis that is essential for its solution in LCA as well. That is that allocating in one process is preceded by the breakdown of that process, through the five-step procedure, into several parts, each with its own operational method of allocation. The function of the subprocess in the broader process remains the central element throughout the procedure. The assignment of the subfunctions differs, however, for each of the parts, as does the practical operation of allocation.

Taking the five step procedure of economists as a starting point, the allocation procedure for LCA may now be described analogously. On a technical basis, all process aspects are first split off from the multiple process that contributes to one type of goods or service only, that is process part 1. These subprocesses are the *first type of constructed single output processes* of the allocation method developed here. Within this part of the process no allocation is required; splitting it off is part of the allocation procedure.
Secondly, all process aspects are separated from the multiple process that contribute the same function to all goods and services the process produces, that is process part 2a. An example of such attributable overheads are in-company services. In some installations their functional contribution may for example be made operational for each product as the share in total running time. These subprocesses are the *second type of constructed single output processes* of the allocation method developed here.

These first two steps split off all process aspects for which the function may be made operational singly, either functional characteristics being used, or physical aspects if these are good indicators of the function. The remaining combined and joint kernel of the process, parts 2b, 2c and 3, are really multiple, for purely technical or for economic reasons. For these purely combined and joint parts of the process the different functions cannot be specified in materially similar terms, by definition, since such process parts have all been split off in the preceding two steps of the procedure. Each of the remaining functions of course has its own physical form, but each is different and thus cannot be used for allocation between these functions. The only common denominator for the purely joint goods and services resulting is their value. That value is not the product value of the final products of the process. It is derived, after the subtraction of all costs incurred for the single subprocesses of step 1 and 2*a*, by means of the gross sales value method. All remaining inputs and outputs, not being the goods or service produced, are allocated to these "intermediate" products, on the basis of only their thus attributed value. Thus the remaining body of the multiple process, encompassing parts 2b, 2c and 3, is split up in the allocation procedure, by the gross sales value method, into a third type of constructed single output processes. It makes no difference whether the allocation is by part or for all three types of parts together. The resulting constructed single output processes cannot function individually.

6. Allocation in the three pure types of multiple processes

This procedure for allocation in LCA would now be applicable to any economic process, single or multiple, with positively valued inputs, or outputs, or both. Even if the process is a single one the allocation procedure may be applied. In that case the first process part contains the full process, the other parts are non-existent. For quantification specification of the function is still required, possibly (but not necessarily) made operational in terms of physical characteristics. The next question is how the general method now developed applies to the three basic types of multiple processes in LCA where the allocation problems occur, now including also the inputs from the environment and the outputs to it that sofar have been neglected.

Applying the method there involves the construction of several single output processes. With cost allocation, the only items to allocate are economic inputs and outputs. With allocation in LCA the allocation of economic inputs and outputs can and should be exactly the same, as indicated above. The question now is how the "split off" allocation procedure for LCA may be applied to environmental inputs and outputs. The application is staged in the three main types of processes where allocation is required for LCA: multi-output combined production processes, multi-input waste handling processes, and one-input-one-output recycling processes.

<u>Multi-output production processes</u>
The first step is to split off those subprocesses that are technically connected to only one product, one type of goods or one service, that is part 1. Such subprocesses require environmental inputs and economic inputs and lead to environmental outputs and negatively valued economic outputs. These inputs and outputs are attributed only to that one product. The product function may be defined by means of physical terms as indicators of the function. What makes this step different from normal process specification in the process tree is that a subprocess thus discerned need not be one that would otherwise have been specified separately. Further treatment is exactly as in single processes, no allocation is involved. The product itself may sometimes flow through quite unchanged. An example is the emissions of a pharmaceutical product at the phase of transport from the separator installation to the packaging machine, in the combined production process of that drug and fodder.

The second step is to split off those parts of the process that are required for each of the products in attributable amounts, parts 2a and 2b. The allocation of both environmental inputs and outputs is exactly the same as for the economic inputs and outputs summarized in figure 4.2.10 above. The allocation is based on the one type of function this process part contributes to both products. In the quantification of the attributable amounts the function of the subprocess is the key element. This function may be made operational through the relative contribution to sales, through the gross sales value method. The function may also be specified in terms of the function concerned, as with the number or total time of telephone calls. In exceptional cases computation can be in terms of physical units such as mass. In this step in the allocation procedure a second type of single process is thus constructed. In that single process the environmental inputs and outputs are treated as in any other single process, by the quantification of the one valued output of that process in some unit. That unit of function may, again, have a physical component, such as transport in ton-kilometres.

For the remaining fully joint parts of the process the gross sales value method is applied to the environmental inputs and outputs in exactly the same way as to the economic inputs and outputs; in this way the last step in the allocation procedure is covered. There is no alternative in terms of physical causation since the units to be allocated to are outputs, and outputs cannot cause inputs in a physical sense. *Physical allocation can never be involved in the multi-output pure type of allocation problem.*

Multi-input waste handling processes

Again, in the first step all process parts are specified that are necessary for the processing of one of the several wastes only, e.g. the extraction of water from kitchen refuse before burning it together with other wastes.

The second step involves splitting off the attributable subprocesses required for more than one specific waste. The operations of a metal separator may be attributed to the several metal containing waste flows, for example, in proportion to the amount of metals in each type of waste. The function might also be made operational through the gross sales value method, through the contribution the waste streams concerned make to total sales value. The processing costs (plus average profits) for this part are then allocated proportionally to the total proceeds of their processing. This method of becoming operational is at least as good as that based on amount of metal by mass. That physical measure, for example does not indicate the electricity use caused in separation, since electricity use is inversely related to the concentration of the metal in the waste.

For the remaining purely joint part of waste processing (2b, 2c and 3), no technical reasoning allows the further differentiation of *economic inputs and outputs* according to the waste causing them. The remaining process is, economically, purely joint[1] by definition. The *environmental inputs* of this joint part of the process cannot be treated other than economically, through the gross sales value method. The question now is whether this is also true for the *environmental outputs*. Here social and physical causation work in the same direction. The final criterion for the analysis is, as always, how the inputs and outputs would be influenced by a slight change in the amount of one input. It is clear that in so far as a change in environmental outputs occurs in a physical causal model, this change is to be attributed to the input in question. A simple example is the cadmium from rechargeable batteries. If that waste flow is the only source of cadmium emissions, they are fully attributable to these batteries. But the analysis will not usually be that simple. Suppose that an emission is caused by the combination of a limited number of wastes. The example is dioxin from burning wastes containing chlorine together with wastes containing carbon. The wastes in the example are PVC, with a high ratio of chlorine to carbon and a small absolute amount of chlorine, and kitchen refuse, with a low ratio of chlorine to carbon but a high absolute amount of chlorine. Let us assume first that a relation exists only between the amount of these two different wastes processed and the amount of dioxin emitted. The amount caused by each waste is then its marginal contribution. Marginal contributions, per unit, multiplied by their amount equals the full amount emitted. Thus the dioxin-forming part of the waste processing might be specified in an analytical physical model, should the technical process data allow such a procedure. Allocation of emissions to the different wastes concerned is thus according to their physical causal contribution of these emissions.

The high chlorine content of PVC suggests that dioxins are caused especially by PVC wastes. It has proved impossible however to find experimental data to support this

[1] Waste processing installations generally do not require one specific mixture of wastes. Consequently, waste processing is not a case of pure joint production, in the economic sense, but of combined production by one installation. Here a breakdown has already been made of the separable subprocesses.

position. The best guess now, let us assume, is that for a given installation[1], the emissions caused are virtually independent of the amount of chlorine wastes or carbon containing wastes in the broad range of waste compositions related to the waste composition actually encountered. Given this non-linearity, the marginal contributions are effectively zero for both waste flows processed and emissions of dioxin occurring cannot be explained physically by the individual waste flows processed. These dioxin emissions are purely of a joint nature. It then is the total of all wastes processed that collectively causes the totality of these emissions. The allocation then can be based only on the economic *raison d'être* of the process, according to the contribution of each waste flow to total proceeds, i.e. according to the gross sales value method.

Thus, for the purely joint part of waste processing, a physical-causal analysis may allocate part of the emissions, the attributable part, to specific wastes processed. Some remaining wastes with negative value, delivered to a subsequent waste processing unit may be attributed on the basis of physical causation in exactly the same way as emissions to the environment. All other emissions and all other inputs and outputs can only be allocated according to the gross sales value method. Applying the method to these remaining emissions and the remaining wastes to be further processed is fully equivalent to the allocation of economic inputs and outputs.

The processing of a negative value waste contributes to gross sales since the supplier of that waste must pay for the service of processing. There are some practical peculiarities here. Payment for waste processing is often to some extent collective. Moreover, payment may not be charged for amounts delivered but for the right to deliver. Sometimes even that connection with payment is lost. Waste processing is then paid on a total cost basis from taxes and fees or similar levies. When no data on value are available, the allocation may be based on attributed costs that in another context would simply have been the prices. For this situation, quite common with household waste, a technical economic model could generate shadow prices for assigning the relevant economic values[2].

<u>One-input-one-output recycling processes</u>[3]
For pure recycling, i.e. the processing of a negatively valued waste into a positively valued product, the procedure is simple. Separate subprocesses or combined overheads might be involved. For the result of the allocation procedure, however, it makes no difference whether they are split off and allocated or all included in the purely joint process part. In both cases the only allocation mechanism applicable is the gross sales value method combined with physical allocation where applicable. Suppose quite realistically, that the recycling process consists of a successive number of subprocesses. If a waste is processed in such a subprocess, that activity takes place because it is necessary either as part of "getting rid" of the waste or as part of making the product to be sold, or

[1] The amount of emissions is mainly dependent on the type of installation. That type is given here. The question remaining then is to which wastes this given amount of dioxin is to be attributed quantitatively. See Born 1992 for a thorough survey and analysis of this tricky subject.

[2] One could combine the technical economic and the physical causal model into one. That model then would specify the allocation factors for the joint part of the process, and may be used for the combined attributable subprocesses.

[3] This only concerns a waste-to-be-processed, with a negative value, that is transformed into a product of positive value. The recycling of wastes with a positive value, where the processor would have to pay for receiving the waste, is fully equivalent to the multi-output production case from an allocation point of view.

both. If this activity were split off and treated independently, the only basis for doing so would be the costs of the process (plus the contribution of that part of activities to total profits). Splitting off the process thus effectively involves the gross sales value method. Adding all processes thus split off would result in the full recycling process again. Applying the method to the full process would thus seem the more straightforward approach.

Two steps are involved, that of physical causation and that of social causation. The first to be split off are those parts of the process that may be explained purely physically. This only involves the contributions of the waste to be processed to only the *non-product outputs*. To be made operational, this step needs a quantified model of the process. Those parts of the non-product outputs that are explained by the amount and composition of the waste to be processed are to attributed to the processed waste only, after correcting for the possible changes in the value (amount times price) of the product produced that occur simultaneously. *If the latter factor were not subtracted the amount explained physically would still be explained by the two products together between which the outputs were to be allocated!* The process parts which remain, analytical, not physical, entities, are now fully joint since neither a functional nor a physical analysis can split off further parts. These process parts can be explained only by the values of the two products (waste processing service + good produced) they realise, through the gross sales value method. Gross sales here are the sum of the proceeds from the service of waste processing and the sale of the goods produced in the recycling process. If no data on values are available, as often the case with government operated installations, a shadow cost pricing procedure is the only possibility to make operational the allocation of this remaining fully joint process element. See figure 4.2.11 for a survey of the operational steps in the general method for allocation in LCA developed here.

When the value based method for allocation was proposed at the SETAC workshop in Leiden in 1991, it was received positively in principle, (see SETAC 1992, p.72). However, several practical arguments were raised indicating that value-based allocation may be difficult in practice. The central problem is how to arrive at prices as the relevant value indicators. The value, quantified in financial terms, e.g. as a price in a market, is based on the function a type of goods or service has for the party acquiring it. These practical arguments relate to the extra data requirements, the less stable nature of economic data, the lack of economic data in many instances, and the less objective nature of economic data. Here follow observations on each.

The extra data required are those for the different values created by the process. In production these may be quantified by proceeds, that is by amount times price. Since only the *relative* share in proceeds must be known, it is relative amounts and relative prices that are required. Data on relative physical quantities are already available, so it is data on prices that have to be added. Such data are not always easy to come by. In the context of the vast amounts of data required on technical aspects of production and on related environmental interferences and their effects, this seems a surmountable problem. In principle, more published data on the prices of the goods and services from processes are available now than on the environmental performance of these processes! It is in multiple processes only that a price should be available for each type of goods or service.

FIGURE 4.2.11 ALLOCATION METHODS FOR THE PURE TYPES OF MULTIPLE PROCESSES IN LCA

MULTIPLE PROCESSES DIVIDED INTO PARTS PER TYPE	*METHOD OF ALLOCATION*
multi-output production processes	
part 1 separate subprocesses: *function related Example: Cleaning of hides in a slaughterhouse	**no allocation involved*, the one function possibly stated partly in physical terms, for quantification
part 2a attributable combined subprocesses: *function related Example: telephone calls	**allocation based on one function*, made operational by gross sales value method or by relevant function
part 2b+2c+3 non-attributable combined subprocesses + joint overheads + joint process kernel: *value related only Examples: passengers and freight in air transport; production of chlorine, sodium hydroxide, and hydrogen	**allocation based on two inseparable functions*, made operational through gross sales value method only, ***no*** *physical allocation involved*
multi-input waste handling processes	
part 1 separate subprocesses: *function related Example: Breaking bottles before their entering into an incinerator for household waste	**no allocation involved,* the one function possibly stated partly in physical terms, for quantification
part 2a attributable combined subprocesses: *function related Example: additional heating in household waste incinerator, taking into account specific heat and water content	**allocation based on one function*, made operational by gross sales value method or by relevant function, partly in physical terms
part 2b+2c+3 non-attributable combined subprocesses + joint overheads + pure joint production I, economic and environmental inputs: *value related Examples: maintenance of installations; functioning of staff bureau and administration	**allocation based on one function*, made operational through gross sales value method
non-attributable combined subprocesses + joint overheads + pure joint production II, attributable economic and environmental outputs: *through physical-causal model Example: cadmium in emissions from cadmium in wastes processed	**allocation based on physical causal model*, as one product separate causation
non-attributable combined subprocesses + joint overheads + pure joint production III, non-attributable economic and environmental outputs: *value based Example: dioxin in emissions[1]	**allocation based on two inseparable functions*, made operational through gross sales value method
one-input-one-output recycling processes	
part 1 splitting off **not relevant** Example: transport and storage of pig manure to be processed; sales department for methane	* see part 3
part 2a splitting ●ff **not relevant** Example: telephone calls of pig manure processor	* see part 3
part 2b+2c+3 pure joint process parts I: *through physical-causal model Example: copper containing wastes from pig manure fermentation	**allocation based on physical causal model*, as one product separate causation
fully joint process parts remaining: *value related Example: pig manure processed into methane	**allocation based on two inseparable functions*, made operational through gross sales value method

[1] Assuming that, in the relevant range, variations in the amount of chlorine introduced into the incinerator do not influence the amount of dioxins emitted.

Data on values in terms of prices are much more subject to changes in time than data on technical aspects, so it is often assumed. It is not changes in absolute level of prices that have to be taken into account. Thus changes due to inflation are irrelevant. For the analysis of current products a time series based on linear regression for the last five years might be used, deflated at the current general price level. This will usually give a reasonably stable figure. For comparisons of future products based on future technologies future relative prices should also be available, predicted or estimated[1]. Both problems also occur in data on emissions. Emission data may be very variable in time and future emissions can only be predicted and estimated. The practical problems connected to the value-based part of allocation do not seem insurmountable. They are very similar to other data requirements in LCA inventory. Many emissions may also be variable in time and may be secret or unknown.

The next problem indicated is that of missing markets. The value of inputs and outputs may only be quantified in market terms, if they are sold in an adequate market. Many collective services do not have a market price, their costs are somehow covered by taxes and fees, as with the processing of household waste. The collective payment for these services may be taken as an indication of their minimum value to society, just as the prices of marketed goods and services indicate their minimum value for those acquiring them privately. It is the costs that are quantified in this manner. Sometimes a firm may supply a very inadequate market, e.g. a monopoly. This would disturb allocation mainly if one of its products would be a monopoly product, the other not. In such very exceptional cases a practical solution may be found ad hoc. In-firm deliveries often go unpriced as well. The gross sales value method designed by economists should then be used for internal accounting. The latter is also required for other reasons, such as for cost allocation. Decentralized decision-making within the firm, as with unit management, requires an in-company specification of costs as well.

Finally, some people regard data on value in monetary terms to be less objective than data on physical aspects of technology and thus prefer to use physical data. From an epistemological point of view this is an untenable position. It is social causation that should be established, with a subrole for physical causation in some instances, not physical causation as an independent analysis. It seems odd to measure a variable with a precise but unsuitable measuring instrument, e.g. colour by meters, because the latter are more precise!

7. A general method for allocation in LCA: results

Now that this quite complex line of reasoning has been covered the results may be condensed into a more general form, stating how allocation is to proceed in any multiple process. Multiple processes have been defined as producing more than one product, type of goods or service. Services may be supplied both at the output end (e.g. transport) and at the input end (waste processing). The general method of allocation for LCA as

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[1] Expected prices are of central importance in investment decisions. The price projections then are available to the party who is making these decisions but usually not to others. The techniques are well known. They range from the simple linear regression on the time series to maximum likelihood estimations in multi-equation dynamic models. See for example Harvey (1990).

developed is based on the analysis of causation: allocation specifies the effects caused by each of the products involved.

Two different types of causation play a role. The *first* type of causation is social. It concerns the *raison d'être* of any economic process, that is the function it creates in other economic processes and the value it therefore has. One central feature of this social type of causation is that it may seemingly move backwards in time, similar to *ultimate causation* in Darwinian biology, where species seem to "adapt" to circumstances. The apparent backward movement in time here is based on the *expectations* actors have of future situations and the motives they assume to be operant in others. More mundanely: a product is produced because the producer expects it to have a function that is valuable to others who for that reason are willing to pay for it. The function as the basis for allocation reflects the central element in the social causation of economic processes as intentional activities. The function is the concrete aim of the process. Thus, the causal chain goes "backwards" in the multi-output production process, with the function produced being the *aim* of the operators of the process. A main basis for the practical allocation procedure developed here is this primarily economic analysis of causation in economic theories on cost allocation. The normalization of the economic cost allocation theory results in a general methodical procedure that is directly applicable in LCA to all elements of the process that are intentional. In that procedure the multiple process is broken down into three groups of subprocesses. This breakdown is based on how independently these process parts contribute to the total function of the process, fully independently, partly independently as specifiable in terms of the contribution of one function, or not independently, being purely joint or collective in this functional sense.

The *second* type of causation is physical in a natural science sense. That type of causation is the same as the *post hoc, propter hoc* conception of Hume, the Kantian a-priori causality, or the *efficient causation* in biology reminiscent of Aristotle. No motives and expectations are involved. This causation can only go *in the same direction as time*. In the allocation of inputs and outputs to the several goods produced in an a multi-output process, physical allocation thus cannot play a role. Outputs can never cause inputs in any sense of physical causation. The two other pure types of multiple process have a function that is related to inputs, partly in the case of recycling and fully for multiple waste processing. There, physical causation may play an independent role.

The general allocation procedure results in the apportionment of all non-product inputs and outputs to each of the products causing them. In the allocation procedure for any type of multiple process, also combinations of the pure types, four steps may be distinguished, each with only one basis for allocation. This general allocation procedure is described on the next page[1].

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[1] The method of allocation developed is closely related to a number of other methodological aspects of LCA, such as the boundary definition of the product system. There, it allows a more precise demarcation of the product system studied. The product tree representing the product system is connected to all other product systems through numerous "sprouts" and "roots". Which are to be included and which not? First, all those processes are to be included that contribute goods or a service. Each such combined production string can be cut off directly where it occurs, including the strings of positively-valued wastes going to other product systems. Their recycling there is not part of the product system studied. All inputs required for a multiple process and all not-valued outputs produced, belong to both product systems, in the amounts indicated by allocation. (continued on the next page)

GENERAL ALLOCATION PROCEDURE FOR MULTIPLE PROCESSES

Step 1	The first group of single subprocesses may be split off as separate real processes. They provide their function to only one of the products of the process and *no allocation is involved*. Now all remaining process parts are together really multiple.
Step 2	A group of subprocesses may be split off each of which contributes one quantifiable function to every product. This process part is also single in reality. In this case *the function is the basis for allocation*. The single function of that real process may be made operational in value terms through the gross sales value method or through function specification. The latter may sometimes be expressed partly in material terms. All remaining process parts together are now joint functionally by definition, but not yet joint physically.
Step 3	The next group of subprocesses that can be split off is not joint in the physical sense, that is where a separate physical causal relation can establish an analytically distinct subprocess. There, *physical causation is the basis for allocation*. Physical causation is defined in terms of the changes of items to be allocated as a consequence of a marginal change in the amount or composition of the product to be allocated to. All remaining (analytic, not real) process parts are now fully joint, both functionally and physically.
Step 4	By definition the purely joint process kernel remaining cannot be allocated according to functional or physical causation. The single reason for the existence of this process part is the total value it creates through its products, as redefined in the procedure. There, *the value generated by each product is the basis for allocation*. It can be made operational through the gross sales value method. There now no are no process parts remaining.

4.2.5 Classification: Outline[1]

Introduction

Life Cycle Analysis (LCA) is a decision support system. It helps to consider the environmental effects in decisions on products. With LCA maturing, the data gathered in the inventory has become so complex that an evaluation can no longer be based on them directly, assuming this were possible in the first place. The inventory ends in a list of inputs from the environment, i.e. resources used, and outputs to the environment, i.e. emissions and other direct influences on the environment. This list is the interference table or inventory table of the product system studied. Typically, a table of several hundreds or even thousands of different environmental interferences will result. The decision to support is the (also) environmentally-based choice between two alternative products or variants of them, in any decision context. Such an interference table is made

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The string of processes handling negatively valued waste is one type of service string. In recycling, the string is to be fully included until it has only positively valued outputs. Where the recycling string starts having a positive value, it is also cut off, fully analogous to the combined production situation.

Plain waste processing has no positively valued economic outputs at all. The string has to be followed "to the grave". Other processes supplying wastes to a multiple waste handling process should not be included; they are cut off directly by allocation. However, all economic inputs required for the functioning of waste processing should be included in the process tree.

[1] This section and the next draw heavily on G. Huppes and J.B. Guinée (1992).

for each alternative. The question addressed in this and the following sections is how, on the basis of these lists, the environmentally preferred alternative is to be derived.

There is a growing global consensus that, following the inventory, two different types of analysis are required for answering this question. One is empirical, specifying the potential effects of emissions, resource use and other influences on the environment in an environmental profile or eco-profile. This is the *classification* component of LCA in the framework used here[1]. The subsequent analysis is normative, describing how two or more product systems can be evaluated environmentally, on the basis of their environmental profiles. In this analysis the profile items, as scores of potential effects in the empirical analysis, become the value attributes on which the evaluation is based. This *evaluation* part of LCA is intended to result in an environmental ranking of the alternatives and will be treated in the next section.

Current approaches

Studies currently conducted, differ considerably in how classification and evaluation are handled. In America, a study on paper vs. polystyrene drinking cups (Hocking 1991) sums up emissions in kilograms. Emissions to air, for example are 22.7 kg per metric ton of paper and 53 kg per metric ton of polystyrene. The largest producer of LCAs in the US does the same, with much caution, see Hunt et al. (1992). Since the sum in kilograms has no specific environmental meaning - sulphur dioxide is added to dioxin - it can hardly play a worthwhile role in classification and evaluation. Without an explicit classification, a ranking of alternatives based on a comparison of their interference tables implies an analysis of all individual substances emitted and resources extracted. When many resources and emissions are involved an overall ranking of alternatives will rarely result from this direct evaluation of the inventory results.

In Europe, a software program developed in cooperation with a large car manufacturer in Sweden is at the other extreme. It translates each resource used and each substance emitted into a single environmental evaluation score, that can be added into a total score for the product, see Steenge (1991). Comparing the summed scores of two alternatives tells directly which is environmentally "better." The inventory table is translated into an evaluation in one step, with the omission of a separate classification step. Nearly as comprehensive is a method developed in Switzerland and implemented in software for the largest Swiss food retailer, see Ahbe et al. (1990) and Braunschweig (1992). Its ranking procedure is more transparent. The emissions of the products are ranked according to the allowable total level of emissions in Switzerland, the 'critical load', and by the share of this total load already taken by existing Swiss emissions. The result is a score for each emission type in "ecopoints". First these may be differentiated between the environmental compartments water and air. Similarly, ecopoints are compiled for final solid waste and energy. These ecopoints are then added up without further weighing to a total evaluation.

The attractiveness of the methods in these software programs lies in the ease with which a comparative evaluation can be produced. Define the products and their related processes,

[1] The draft SETAC Portugal framework puts the two elements together in one component, the impact assessment, and distinguishes three steps within this component, the first two (classification and characterisation) covering classification here and the third one, valuation, here being called evaluation.

add some data, and the computer gives the environmental evaluation. Software developed at CML-Leiden, for a study on window frames (see Hoefnagels et al. 1992 on the study and Guinée et al. 1992 for a description of the software), computes ten separate environmental effects in the classification, together forming the environmental profile of a product. The effects distinguished and the primary method to make them operational are the as follows:

* depletion of resources	→as a ratio to the stock of the resource
* depletion of fossil and fissile energy	→in MJ
* acidification	→in units of acidified soil (Dutch norms)
* global warming	→in kg CO^2 equivalents
* ozone layer depletion	→in kg CFK-11 equivalents
* health aspects of air pollution	→in units of polluted air, based on MAC-values
* health and ecosystem effects of water pollution	→in (mainly Dutch) units of polluted water, water quality norms
* final waste, hazardous	→in kg, specified on a list
* final waste, non-haz.	→in kg, all other
* ecosystem degradation	→in terms of the use of wood from rain forests

This Dutch approach to classification does not give an evaluation between these environmental profile items resulting. Many studies in Europe used similar approaches, but with fewer effects in the environmental profile, see Assies (1992b) for a general survey. This hitherto European tradition started in Switzerland (EMPA 1984) and in the Netherlands (Druijff 1984).

Further developments may be expected along these lines, since not only the Leiden workshop of SETAC but also the Sandestin workshop of SETAC, in the US, favour this approach. In the US it has not been applied yet in case studies. Most practitioners, on either side of the Atlantic, now want the steps leading to an evaluation identifiable, with facts and values separated into classification and evaluation respectively.

Two quite similar approaches to classification were presented in the Leiden workshop, each specifying several environmental problems a product may contribute to, see Guinée (1992) and Baumann et al. (1992). In the SETAC-workshop in Sandestin, certain items were formulated slightly differently. The list in table 4.2.12 below is based on Guinée, with one item added from Baumann et al., some items reformulated, based on the preliminary results of the Florida workshop, and two additional items: salination and depletion of global gene stocks. This list nearly fully coincides with a list on possible headings for classification of the LCA-methods project for the Dutch government, see Heijungs et al. 1992b. The items are ordered according to type of problems (depletion, pollution and disturbances) and the spatial level at which they tend to occur (global, continental, regional, local). Not all existing environmental problems can be attributed to products, however. The specific location of a highway i.e. whether or not it cuts through a nature area, is an example. The list of problem items can be extended by any new problems coming up.

FIGURE 4.2.12 LIST OF POSSIBLE ITEMS IN THE ENVIRONMENTAL PROFILE THE RESULT OF CLASSIFICATION

PROBLEM TYPE	PROFILE ITEM: Problem or process mechanisms	SCALE LEVEL OF PROBLEM[1]
Depletion (5)	*depletion of non-renewable resources, energy carriers	global
	*depletion of non-renewable resources, others, esp. minerals	global
	*depletion of renewable resources	global
	*depletion of gene stocks	global
	*space requirements	regional
Pollution (15)	*global warming	global
	*ozone layer depletion	global
	*human toxicity	continental
	*ecotoxicity	continental
	*acidification	continental
	*ionizing radiation, including emissions of unstable isotopes	continental
	*photo-oxidant formation	regional
	*nutrification	regional
	*salination	local
	*heat	local
	*noise	local
	*smell	local
	*occupational health	local
	*consumer health	local
	*final solid waste	local
Disturbances (7)	*drought/desiccation	regional
	*landscape degradation	regional
	*ecosystem degradation	regional
	*external safety	regional
	*visual disturbance	local
	*occupational safety	local
	*consumer safety	local

A taxonomy of environmental classifications

In the Sandestin workshop, a general discussion was held on methods for classification. On the one hand, there were many people with a background in Environmental Impact Assessment (EIA), which is primarily used for evaluating a single large investment project at one specific location against alternatives to it. These practitioners are accustomed to approaching the environmental analysis on a site-specific basis, where the following type of descriptive analysis is common. "This installation here harms a nature area over there and, if it explodes, all those people living over there on the other side of the road might be injured or killed." This approach contrasts with the approach to classification in LCA, that taken in Europe which computes for example, "critical volumes of air" showing an amount of air that might be polluted to an unacceptable level, as indicated by MAC-values. Location aspects, concentrations, duration of exposure, and number of people exposed are all abstracted from the data. The resulting effect scores indicate potential effects; they are a *measure of concern*, expressed in physical entities.

[1] These scale levels are feasible, see below, and might be chosen differently.

The current American approach in case studies thus far generally leaves out any formalized classification steps and analyzes the emissions directly into an evaluation. A change appears to be under way, however. During the discussions in Sandestin (no final report available yet) four optional types of impact analysis were proposed:

Loading analysis, indicating possible negative effects of the emissions and, more generally, of all environmental interferences. This is the current American approach in LCA case studies.

Equivalent unit analysis, as in the example of global warming. This is the most current European approach in LCA, in which a "real" effect is not modelled but only an indication of the potential seriousness, the level of concern, is quantified.

Generic exposure/effect analysis, based on expected concentrations. This approach has not been applied to products. Models incorporating types of emission site, emission dispersion and transformation, and immission concentration are required. Such an analysis seems more applicable to the analysis of single processes, such as waste incineration, or of single hazardous substances, such as cadmium, and less to products. In the inventory, the type of location, at the scale level distinguished in the models, would have to be specified.

Site-specific analysis of effects of exposure, also based on concentrations. This approach stems from Environmental Impact Assessment. Real effects, in the real circumstances where they take place, are predicted. In this approach, product systems have to be specified in the inventory according to the real locations of their processes.[1]

Of those present in Florida in February 1992, predominantly Americans but also Europeans, many thought the equivalent unit analysis to be the most suitable option for LCA at this stage. This is a first step to a practical convergence between Europe and the U.S.

Basic dimensions in classification

There are at least three principal dimensions that determine the complexity of LCA classification:

◇ the spatial scale on which the effect chains in the environment manifest themselves

◇ the spatial scale on which the effect chains in the environment might be modelled

◇ the spatial scale on which a problem is created, that is the spatial scale of economic causation.

The effect chain of sulphur emissions is very long due to atmospheric transport mechanism. The effect chain of ammonia emissions is much shorter as far as transport is concerned, but the problem in Northwestern Europe has a very similar spatial scale level; it is as widespread, being created on many places. This dimension does not seem to be fundamental for LCA. Of course, if a problem occurs locally at only a few locations, it will hardly be suitable for inclusion in LCA.

When the functional unit of a product is defined, questions can be asked at different geographical levels, comparing, e.g., "European refrigerators, used in the Netherlands". Part or all of the inventory would then be specified for these geographical areas. This might influence choices in classification. Thus a related third element is

◇ the spatial scale at which the product system is specified.

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[1] For a site specific analysis and evaluation of environmental effects, general categories are required as well. See Udo de Haes, Nip and Klijn 1991 for a detailed introduction on this subject.

Wide ranges of combinations are possible in principle. First, Here are two examples that vary the choices in the spatial scale of a problem and the spatial scale of the effect specification. The spatial level of the acidification problem is semi-continental, with large areas of Northwestern and Central Europe affected by any given source. Nevertheless, the effects may be specified, with Dutch sandy soils or Scandinavian lakes taken as a local scale model for specifying effects and detailing transport models to that end. Effects may even be specified in the most real, fully site-specific, terms. Conversely, it is also possible to specify the effects of a local emission related to a local problem based on a global model that abstracts from any differentiation occurring in the world. An example is the health effect of emissions of a substance such as cadmium, that is assessed in the classification in ADI-values of the World Health Organization only. This procedure abstracts from all other exposures, the numbers in the population exposed, differences in eating habits within and between populations, etc. A reasoned choice as to these scale levels should be made if LCAs carried out by different practitioners are to remain comparable.

FIGURE 4.2.13 BASIC DIMENSIONS IN CLASSIFICATION, PER PROBLEM TYPE

scale level in goal definition / inventory:	**scale level of problem mechanism:**	**scale levels of problem modelling in classification:**
global	global (ozone layer depletion)	global (global climate model; potential health effects of toxic substances)
continental	(semi-)continental (acidification)	continental (continentally differentiated climate model)
.....	.....	.....
national	national / regional (over-nutrification)	regional, not yet local / site- specific (effect predictions of global warming also for types of nature areas)
local	local (noise, health effect of toxic substances)	local = site-specific (effect prediction of global warming also for all specific nature areas involved)

Next, the questions asked in a specific LCA, as specified in its goal definition, may differ in spatial scale. On the one extreme there is the general question if, at a global level, phosphate-free washing powders are to be preferred to phosphate containing products. On the other extreme is the question as to what is best environmentally if I, living in a town with a new 96% effective dephosphating installation in my sewage system, with phosphates recycled, should buy phosphate-free brand X, produced at locations a, b and c, or phosphate-containing brand Y, produced at locations k, l and m. Both questions are quite legitimate. Each could be answered, in principle, by either a global or a local inventory analysis, the latter specified at the local level, with so much aluminium produced here and so much there, etc. The classification analysis, similarly, might be based on average or on local concentrations of generally or locally defined problems, modelled globally, or covering geographic characteristics at some spatial scale level. Is the "best" set of classification dimensions that for the lowest scale level that can be distinguished, giving the most realistic description of facts? If the answer were "yes", the complexity of varying classification terms would become unmanageable[1]. Each case study would have to develop its own environmental profile, modelling all its case specific problem items. Thus, at least for the time being, the search is for one standard environmental profile that may reasonably cover many situations.

The question now is which basic type of classification, in terms of the dimensions stated, is best for the present. Let us first eliminate some non-feasible combinations. Luckily, not all combinations of dimension levels are sensibly possible feasible. For example, if the level of specification of the effect chain in the classification model divides the world into one hundred inflow units, the inventory required would have to be differentiated at least according to these hundred geographical units as well. However, if the inventory is on a global scale, it still might make sense to differentiate effect chains at a lower level, to give a better picture of the effects of "spatially average" acidification, for example.

One might search for the truest classification possible. Answering the global question with locally differentiated analyses in terms of effects on concentrations may be truer, in an empirical sense, than a global equivalent-unit approach. For global problems this might be true. For lower level problems the analysis would soon become too unrealistic if the inventory were not specified at least at the level of the environmental mechanisms of these problems. The model would cover specifications that the inventory does not support. The number of criteria on which to classify the emission of one substance might be differentiated into the several thousands types of locations where its effects differ, each with its own specific characteristics. One might even specify all real effects as they evolve in time. However true this detail might be empirically, it would turn both the inventory and the classification into a job taking decades, and it would make an evaluation virtually impossible. The truest choice is simply not workable, for primarily the same reason that extreme utilitarianism is not possible in policy evaluation.

The choice that now can be made probably is between an LCA giving a reasonable idea of potential environmental damages caused by a product, and nothing at all. The ideal of "really" knowing every related effect will not be achievable for a long time to come. If it were applied, it would lead to extreme problems in evaluation (even if formalized, see below). How could this morass with its lethal lurking dangers for LCA be avoided or drained? The first option, avoiding the morass by skipping the classification altogether, has been the American approach until recently. It appears, however, that the increasingly complex results of the inventory cannot be used for evaluation either, at least not rationally. In the example of paper cups versus polystyrene cups, the evaluation is that "it would appear that polystyrene foam cups should be given a much more even handed assessment as regards their environmental impact relative to paper cups" (Hocking 1991). Even this very limited statement is based on adding highly differing hazardous substances by mass. If more data on resource use, emissions and other impacts become available, the possibilities for a rational evaluation will further diminish. The second option, draining the morass, has the greater potential.

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[1] Adding one or two items to the environmental profile is easy by comparison.

Preliminary choices
Three preliminary choices could help define a workable "base line" for classification:
◇ only supra-local problems are considered
◇ effect specification is always globally uniform, even if the inventory is defined more locally
◇ the consecutive evaluation is formalized.

The arguments for the proposed choices are primarily negative. If these choices are made differently, the practical possibilities for life cycle analysis remain limited. Its role in a macro policy instrument would then remain limited as well. The consideration of local problems would add a host of criteria to the classification that demand data for the inventory and complicate the analysis in the evaluation. When this base line has been established, complexity can be added along the two dimensions. First, more local problems may be added to the profile, such as the health and safety of workers. Secondly, the effect analysis could be differentiated between types of locations, with a separate problem score for each location.

Leaving the global level in inflow specification for the classification models would require the specification of the locations of all processes related to the products investigated. This is a tedious task in LCA inventory. No tag can be glued to a plastic bag specifying the oil wells the material comes from. Assuming this procedure makes sense at all the oil wells and other resources that have been used for energy generation, several capital goods, transport, etc., would have to be specified on the tag as well. One could imagine a step halfway between, with effect specifications at a continental level. This option could be made manageable in the not-too-distant future. It would still not make much sense, however, for products traded in more or less homogeneous global markets, like for most metals.

If the evaluation is not formalized, there generally will not be a preferred alternative and no decision support can be given. This limitation would effectively enfeeble the instrument built on it. Without a formalized evaluation, the number of valued attributes in the environmental profile, as given in table 1, should be pruned extensively to not more than seven for general use in LCA and perhaps a few more if an expert panel makes the assessment. If the evaluation is formalized, a much higher number becomes manageable. The twenty-seven items suggested seem too numerous, however.

Through these choices, the gross list of problem items in figure 4.2.12 is reduced for the time being with the exclusion of the items that cannot be attributed to a generally defined functional unit, or cannot be made operational at a global level. All these provisional exclusions are problems with a local scale of environmental mechanisms: heat, noise, smell, visual disturbance, salification, occupational health, occupational safety, consumer health, and consumer safety.

There are two items that do not fit into this problem oriented scheme but seem too important to be omitted. First, the storage of *final solid waste*, a process of the process tree in the inventory, cannot yet be expressed in terms of its impacts on the environment. In principle these are the usual inputs from and outputs to the environment that are specified for any process, e.g. its emissions of harmful substances to air through degassing and

wind erosion, to water through run-offs, and to soil and ground water through leakage. A temporary but practical solution is the treatment, as by Hoefnagels et al. (1992), in terms of kilograms of "waste" and "hazardous waste". In due time the underlying emissions will be treated as any other emissions to water, air and soil.
Secondly, it is not yet possible in the classification to translate *ionizing radiation*, directly emitted or through emissions of unstable isotopes, into health effects in a manner comparable to that used for the toxicity of substances. Radiation plays a role in energy production, as in the case of uranium in coal and several wastes from nuclear energy production. Building materials may also be a source of occupational and consumer exposures to ionizing radiation.
These two items, of another nature than substance extractions and emissions, should somehow be included in the environmental profile until a comparable classification procedure has been worked out for them. See figure 4.2.14 for the resulting reduced list of profile items.

The preferred approach for setting up classification models for each problem type would have the following structure. First, each interference - emission, resource use or disturbance - would be transformed by a factor giving the long-term probability that the direct contribution to any of the given list of problems will occur. With emissions, this factor would be based on some type of multi-media world model, with one or more receptors for each problem type, as with MacKay models. This "extinction factor" or "route factor" leads to a purely cross-media approach in the further effect analysis.
Secondly, the contribution to each problem type is computed in a factor combining all receptor mechanisms of a given problem, the "effect factor" or "receptor factor". Health effects of a heavy metal, in terms of kidney problems or reduced immune response, would be combined, for example into an Acceptable Daily Intake, as is currently being done for many substances by the WHO. The route to intake, as specified in the first factor, would here include environmental transport, food chains, and inhalation. The contribution to potential human health effects of a certain emission would thus be "kg person days of acceptable daily intake". These problem contributions could next be specified in terms of an index, as an *equivalent unit*, with the problem contributions of one

FIGURE 4.2.14 ITEMS IN THE ENVIRONMENTAL PROFILE FOR BASIC LCA

<table>
<tr><th colspan="2">PROFILE ITEMS (VALUE ATTRIBUTES)</th></tr>
<tr><td>Emission effects on:
* climate change
* ozone depletion
* human toxicity
* ecotoxicity
* acidification
* photo-oxidant formation
* over-nutrification</td><td>Depletion of:
* non-renewable resources, energy carriers
* non-renewable resources, minerals
* non-renewable resources, gene stocks
* renewable resources
* land space for ecosystems</td></tr>
<tr><td>Disturbances leading to:
* external (un)safety
* ecosystem degradation</td><td>Miscellaneous:
* landfill
* radiation</td></tr>
</table>

substance as the basis, as with CO_2 in global warming. For some problems, the measures for the extinction factor and receptor factor are available. Ozone layer depletion is an example. See for a specification of this approach for toxic substances Guinée and Heijungs (1993b) and for a more general treatment of this subject Guinée et al. (1993b). More progress on this and other classification models may be expected in the near future. The classification models are not the usual empirical models. They do not represent an existing reality. They cannot be calibrated through empirical measurements. These models indicate potential effects only. However, they are incredibly more meaningful in comparison to the specification of potential health effects in terms of "summed mass of hazardous emissions in kg".

Normalization of problem scores

The classification translates interferences of a product system into their problem contributions. The partial modelling used rules out attaching any specific meaning to the numbers resulting. There is one final step, however, that could add to the meaning of the classification results, i.e. the normalization of the problem scores in terms of their relative contributions to the problems specified. The following two transformations might be carried out.

First, all the interferences contributing to the problem in question for a year for example, could be transformed into that year's "total problem creation", e.g. all potential ozone layer depleting emissions could be transformed into the total ozone depletion score for that year. The choices of the one year period and of a specific year are quite arbitrary. The year total computed for each problem type indicates an average level of problem contributions for all functional units of the total volume of (final) consumption in that year. The year chosen, year i, should be as relevant as possible, e.g. the last year on which data is available. The potential ozone layer depletion caused by one functional unit of the product analyzed may now be expressed as a fraction of that year's total contribution to ozone layer depletion, using the same model to transform emissions into problems. Thus a new variable is generated:
"*the fraction of the total global warming problem in year$_i$ caused by one functional unit*", with the dimension [*year*].

A separate decision is required here on what constitutes "the total problem in year i". It might be defined at a global level, with the aggregation of all interferences at that level into all problems they contribute to. If the product systems analysed function at a regional level only, one might consider a normalisation at that regional level[1]. The consequence is that the importance of the now regionalised problem is smaller, it is a part of the world problem only, but the relative contribution of the functional unit to this regional problem becomes greater. In most cases, the relevant level would be global as long as the inventory is specified on a global level and concerns the global level.

The fraction ("percentage") results by division by a factor that is a constant in any comparison between alternatives. This division thus leaves any relative scores between

[1] In an LCA model of NAM, for ranking emission reducing investments, normalisation has partly been done at a European level, see Wit, Taselaar, Heijungs and Huppes (1993). NAM is the main Dutch natural gas producer.

alternatives unaffected for each profile item. Thus there is no alteration of meaning for each profile item. However, this normalization allows an actual comparison of how high (not how important) the relative contribution of the product to different problems is. The normalized score of product A indicates how much its contribution to global warming is compared to its contribution to e.g. acidification, by a direct comparison of numbers. The absolute scores remain meaningless as the choice of the unit for the functional unit of product is arbitrary. On may take 1 litre of milk consumed, a thousand litres or the actual consumption of an average person a year. Due to the linear nature of the models involved, the amounts in the inventory table, the environmental profile and the normalised environmental profile would vary proportionally. Thus, any multiplicative transformation is allowed since it does not alter the meaning of results[1].

Secondly, in a transformation that also conserves the full meaning but only applicable at the case level, the highest problem score for any of the alternatives considered is set at 100. The other problem scores, all lower, may then be expressed as percentages of relative problem contribution. The "quantitative importance of a problem score" may thus be measured at a ratio scale level, e.g., the contribution of product A to global warming is 3.35 times as high as its contribution to the acidification problem. In the classification, no comparison of the relative normative importance of the total level of the problems is given. That is a subject for evaluation. The quantification of the classification should not be interpreted as evaluative, indicating for example that improvements in the highest problem scores are "more important" than improvements in problems with a lower score. That normative judgement can only be based on an evaluation of the relative normative importance of the problems concerned.

Conclusions

There is a convergence between LCA theorists on both sides of the Atlantic on the environmental analysis in LCA. A separate classification (= SETAC classification plus characterization) is to be included in LCAs. It describes the potential contribution of a product to several environmental problems, i.e. its environmental profile. For each profile item a model translates all relevant interferences into a non-neutral receptor mechanism, "the problem". The main structure of these potential effect models is that they at least include an extinction factor and a receptor mechanism. There also is a growing consensus on the types of problems that might be included in the environmental profile, including at least climate change, ozone layer depletion, acidification and human toxicity.

For the time being, this classification can be based only on a globally uniform environmental effect analysis. A gathering list of effects currently specifies twenty-seven items for the environmental profile of a product, of which about a dozen have now been made operational, in now still widely differing manners. If no formalized evaluation becomes available, this number should be reduced to about seven. Specialized committees might handle up to ten items. A formalized procedure could handle many more. The most

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[1] Any such linear transformation is allowed. One main reason for this state of affairs is the fully arbitrary choice of the quantity of the functional unit. 'Drinking 1 litre of milk', 'drinking 1000 litres of milk' and 'drinking 1.000.000 litres of milk' are fully equivalent functional units. The problem score still is a ratio scale variable, with both a meaningful zero point (no problem contribution) and interval (the contribution of process X can be added to the contribution of process Y) defined, given a well defined functional unit.

practical proposal, at least for the short term, is to limit the classification to a global analysis of non-local environmental problems. A list of sixteen profile items results, see table 4.2.14 above. Several of these environmental profile items are currently being made operational.

Normalizing the profile scores allows a comparison of products on their relative contribution to different problems, e.g. much above average or somewhat below. No comparison of the importance of scores is possible for allocation since a normative judgement on the importance of problems is required.

In setting up a standard method for LCA, governments would have to specify a minimum number of classification items. As in the case of global warming and ozone depletion, governments could effect international consensus on the classification models to be used. This would constitute one very important step towards creating the authoritative LCA society-environment interface.

4.2.6 Evaluation: weighing of problems

Introduction

The evaluation is the final component that LCA, as a decision support tool, is all about. If the tool is viewed at the policy instrument level, the question in the evaluation is whether the environmental target is to be defined as the contribution of "all interferences to all problems" (that is target level h* in table 2.3.3 in Part Two) or in terms of the "total environmental effect" (target level j* in the same table). As indicated, the comparative analysis of product systems will rarely lead to clear cut results if the evaluation is restricted to using the problem items of the environmental profile as the sole value attributes. Only if one alternative is fully dominant over another can a well-substantiated environmental ranking of alternatives result[1]. However, even minor changes in product systems will usually have divergent effects on different problem scores. The use of the instrument would remain limited to very obvious improvements such as using less material, while leaving the function of the products fully unchanged. For such trivial cases of dominance a quantified LCA does not make sense. Even such simple improvements as increasing the fuel efficiency of cars through better maintenance, will not usually be dominant. In the car example, first more maintenance is required, and secondly higher efficiency will usually lead to larger emissions of NO_x. Supposing that an effective classification has become operational (establishing level h*), *the central question for this section is how an overall environmental evaluation might be constructed* (creating level j*).

That subject is the domain of decision analysis. If the overall evaluation were not built into the instrument, overall evaluations might still result in individual cases in an *ad hoc* ranking procedure. These cannot be authoritative for society as a whole, being inconsistent for different groups making choices and between choices on different product

[1] If an alternative is dominant at the level of the environmental profile it does not have to be dominant at the level of the interference table. The reverse is always true however. Thus, if an alternative is dominant at the interference level it is the preferred alternative, **independent** of the methods of classification and **independent** of priorities that might be assigned to problems. Dominance at the level of the interference table is even more improbable than at the level of the environmental profile.

alternatives. If such inconsistencies between evaluations did not occur, (if society as a whole would behave rationally in a von Neumann sense, see von Neumann and Morgenstern 1944) there would exist at least one set of weights that can produce the same evaluations in a formalized manner. That set of weights would be the revealed societal preference.

The odds against consistency developing spontaneously are high, especially if a start with some authoritative system of weighting is not made. This section further builds on the general decision-making theory to be incorporated into the standard methodology for LCA.

Decision analysis

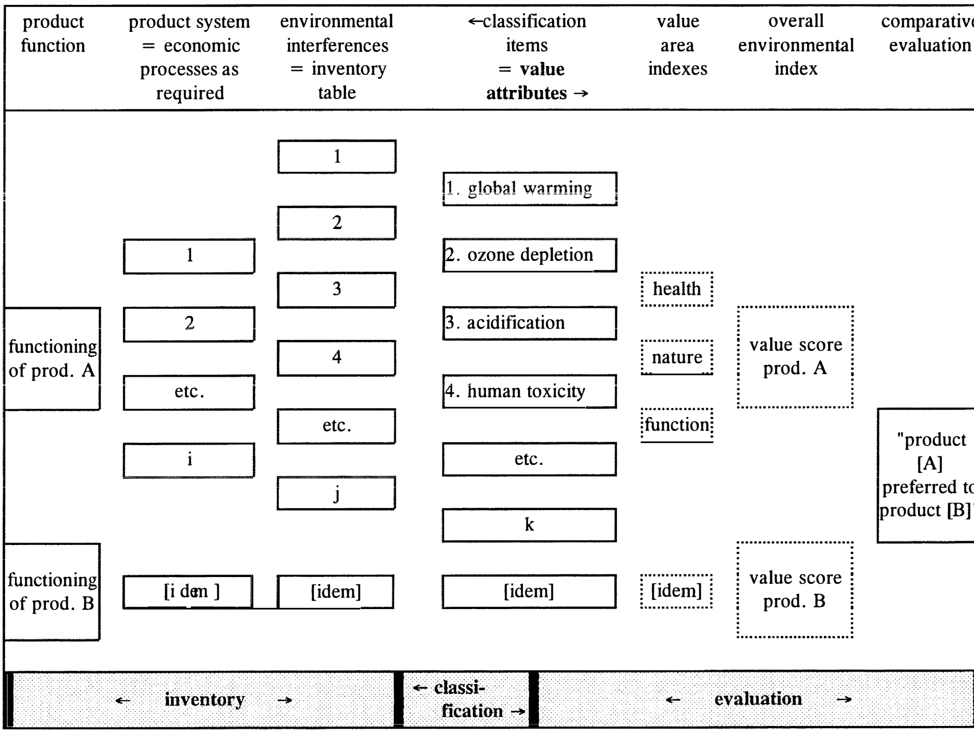
Formal decision analysis for decision support is an old discipline developed at the beginning of the eighteenth century by the Swiss Bernoulli brothers, Niclaus and Daniel. See Bell et al. 1988, especially the survey in chapter 1. As Edwards et al. (1988, p.459) point out:

> *"Decision analysis is almost completely concerned with instrumental acts producing instrumental values. ... The consequences of an instrumental act derive whatever value they may have from later acts and consequences to which they may lead."*

Each consequence itself may be the node of several other consequences. These may be valued because of subsequent consequences, etc. At the end of each sequence, however, the last item must be a valued one. Without it, the effect is neutral and thus irrelevant to the evaluation analysis. The effects specified should be as close to values as possible, usually requiring the complex long-term effect chain. Clearly there is a trade-off between simplicity and completeness when constructing a consequence - value tree. Either sprouts have to be pruned, that is lines of consequences have to be disregarded, which makes the analysis a partial one, or the consecutive number of consequences has to be kept low, restricting the analysis to direct, often short-term, effects, which makes a connection to values more difficult. Still easier of course, is the analysis of only direct and partial effects. It is also less meaningful.

The general picture in the analysis for decision support is that for the several alternatives and variants studied, the tree first branches out in terms of increasingly complex consequences. This, at the other end of the picture, condenses again into a few values or even a single value, see figure 4.2.15. The connecting elements between the mainly "outbranching" *consequence tree* and the "inbranching" *value tree* are the *value attributes*. These value attributes are specified in terms of empirical consequences. For LCA, this translates as follows. The result of the classification, the environmental profile, specifies the value attributes that are the basis for the evaluation. The specification of the environmental consequence tree is the subject of the classification and also of the preceding inventory analysis. The specification of how value attributes are related to one or more 'end values' is the subject of the evaluation. Three independent environmental value areas

FIGURE 4.2.15 CONSEQUENCE TREE, VALUE ATTRIBUTES, AND VALUE TREE FOR PRODUCT SYSTEMS A AND B



areas may be distinguished, as suggested in the last two SETAC workshops and also by Udo de Haes (1992):

* human health and physical well-being
* ecosystem quality, as a value per se[1]
* material functions, e.g. resources, also for future generations.

These value areas may function as end values or goal variables. At this point, however, the next step could be the integration of the three value areas into an overall environmental index. Only this latter his step would usually lead to a priority ranking of alternatives in the comparative evaluation.

Evaluation: formalized or not

If the evaluation part of the tree is not formalized but *ad hoc*, the evaluation analysis executed by practitioners and users of LCA seems to have a maximum complexity of about seven independent value attributes. Adding more would lead only to more arbitrary results, not to more well-founded results. In certain public procedures, such as a decision on the right of a producer to use an ecolabel, a trained group of representative specialists

[1] These may also be seen as immaterial functions.

might take into account a somewhat larger number of independent values. Even with seven values, however, it may be expected that results will emerge from
the procedure that differ widely between analysts and practitioners, and between different representative groups of experts. Generally, reasonable conditions set on evaluations by one party will not be met. Such conditions, incompletely drawn from Sen (1969) are as follows:

- ◇ the exclusion of cyclicity (A preferred to B, B to C, and C to A)
- ◇ independence of irrelevant alternatives (result of comparing A and B as candidates for prime choice influenced by the presence of a non-prime choice alternative C)
- ◇ non-dictatorship (one value is always decisive for the outcome, disregarding any quantitative differences in this and all other values).

Any procedure with quantified weights will satisfy these conditions.

If the evaluation part of the tree is formalized, many more value attributes can be considered. The group specifying the evaluation weights can be introduced to the model, they can try out several sets of weights and they can go through an extensive set of sensitivity analyses to exclude any of the flaws unintentionally introduced in the evaluation system. In this way the formal system and individual intuition can approach each other. In such a procedure, it is not necessary that value attributes are fully independent empirically. This requirement, stated above for ad hoc evaluation to reduce complexity in evaluation, is difficult to meet in practice. Emissions of CFCs, for example, deplete the ozone layer, which itself leads to an indirect climate effect. But these CFCs also affect climate directly. "Double counting" can easily be prevented by the use of a hierarchical evaluation procedure, see Saaty (1992). The evaluation group then finally can come up with a set of weights, suitable for routine application to any LCA case study. Experience in quite complicated cases has shown that groups, after some rounds of discussions and adjustments, usually agree to an astonishing extent on the weights given to value attributes and their groupings, see Saaty (1992) and more extensively, Edwards et al. (1988). With such formalized evaluations the manageable number of items in the environmental profile may easily rise to fifteen or twenty.

The boundary between consequence tree and value tree

The value tree and the consequence tree should meet: value attributes not specified in the classification can be omitted from the evaluation set up, and value attributes not further valued can be dropped from the environmental profile and the preceding inventory analysis. The value tree has prevalence over the consequence tree in principle. "The potential environmental damage" is the basis for the comparison of different products and variants and anything relevant should be included. Going from left to right, see figure 4.2.15, the value tree preferably ends with a comparative evaluation of products. Going from the right to the left, aggregate values can be broken down and the resulting sub-values can be broken down again, etc., until the value attribute coincides with a specified consequence as a classification factor in the environmental profile, see the middle column in that figure. It should be noted that the boundary between classification and evaluation is not strictly delimited. Suppose that a valued attribute in the environmental profile is valued because of its assumed contribution to the next consequence. Acidification is valued negatively, for example, since it causes "dead lakes" (a), reduced tree vitality (b), corrosion of concrete and metal objects (c) and respiratory problems (d). These consequences could then be taken out of the evaluation part of the analysis to replace the former "acidification" attribute in the environmental profile. The most reasonable boundary, however, is for the effects that still can be scientifically quantified to belong to the classification[1], while those that cannot, are put in the evaluation. They might be quantified in the latter, on the basis of subjective probabilities and next be assessed on the basis of subjective values. In terms of a decision analyst, the situation of *risk* is to be treated *in the classification* and also *in the inventory*, of course, where effects are certain or the chances of effects are known. The situation of *uncertainty* in valued outcomes, a central problem in decision-making theory, is handled *in the evaluation*, together with all purely normative aspects. Transparency would be increased if these factors could be specified separately. That partly philosophical task lies ahead.

Combined consequence/value tree
Products may, through the economic/environmental route and the value route, have an influence on each value area. At least for the time being, there is no aggregate measure for separate application to these three value nor for the overall environmental index. Monetary valuation (e.g. "willingness-to-pay") breaks down at problems like time preference, dependency on income, and practical measurement. Nor does the costs-of-reduction-of-effects approach make sense in evaluation. Costs for emission reduction of NO_x, for example, differ substantially from those of SO_2 for equal amounts of acidification reduction. In many cases, such as process/product integrated emission reductions, these costs cannot even be established with any degree of precision. The costs induced in actual substance related environmental policies could be used to establish a set of weights for substances. Such a revealed preference approach would not be effective here for two reasons. First, actual policies do not take only environmental aspects into account but also economic aspects ("which burden can they carry") and political aspects ("how does this affect my political constituency"). Secondly, the problem is that no overall weighing procedure exists now. Current policies will thus be inconsistent. The question here is how such an overall weighing procedure and such an overall environmental index is to be established.

The establishment of evaluation weights cannot but rely on the combination of subjective probabilities assigned to further outcomes in the value tree and on the subjective, but may be structured, evaluations of these outcomes. Technically, a formalized weighing procedure requires nothing more than a set of weights for each of the problem items in the environmental profile, that translates these into the "total problem contribution" of the product system studied. In the construction of these weights some intermediate steps may be useful for conceptual clarification, e.g. first setting up weights for the three main value areas, and then for the importance of the three value areas relative to each other.

Edwards et al. (1988, pp.445 ef) describe a general procedure for evaluation. After having established the value tree in their first two steps, their third step is to quantify all attributes in value terms. In LCA this would be the creation of the environmental profile in the inventory and consecutive classification. In step number four these first could be weighted and added into subtotals, e.g. value area scores, and then into the overall

[1] The fact that no classificatory step is possible for a certain interference, as is the case with radiation, does not imply that it should be left out of the evaluation.

environmental index by the weighted addition of the value area scores for each of the alternatives to be compared[1]. Their step five is a general sensitivity analysis. It allows the assessment of the robustness of the results due to all possible empirical and normative variations[2]. The last step six is no longer evaluative decision support, but the decision itself. In decisions on products, other types of normative data than those regarding the environment will nearly always play an independent role. Heijungs et al. (1992b) prescribe similar steps as an optional path for the evaluation. Those objecting to formalized weighing procedures should keep in mind that any set of emission taxes fully implies a set of normative weights, see the next chapter, and that the lack of weights, while decisions still will go on to be made, will inevitably lead to inconsistency at a societal level. The availability of a set of weights, with their accompanying imperfections, does not imply that they should be used blindly.

4.2.7 Cultural working mechanism: the public availability of a standard method and tools for LCA

Life cycle analyses may be applied to many purposes, ranging from forbidding or allowing products on the market by governments, to designing and marketing products by firms and helping decide on the purchase of a given product by individuals or organizations. In the creation of macro-instruments, the application of prohibiting policies to a specific product are hardly worthwhile. An instrument such as ecolabelling also is also intended for application at the level of individual products. Such micro applications of the LCA interface would be tedious since there are so many existing products and so many new products introduced each year. Only a very broad application of such micro-instruments could result in a general working mechanism for environmentally oriented choices. In chapter 1 the choice was made, in principle, of the higher level macro instrument of standardized LCA methodology. *The aim of this section is a more precise description of the government-society interface of that macro LCA instrument.*

It would be most attractive if all choices on products in society were oriented towards the environmentally better alternatives. Most choices affecting products are made privately. They cannot generally be supported by public analysis on a case-by-case basis. Primary examples are in product design and redesign and in strategies for product development. Nevertheless, it is mainly these case-specific choices that affect the environment. The first requirement for a working mechanism covering these choices is that the life cycle analysis, discriminating between environmentally good and bad (or bad and worse) is authoritatively provided and practically available, as information not on cases but on methodology. The second requirement is that there is an incentive, here a cultural one, to make the right choice. The administrative tasks involved in making this instrument operational might relate to both these elements, the information and the incentive.

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[1] Since the transformation of each interference into its contribution to the classification problems is also by a scalar, it is possible technically to rank each individual interference according its 'total problem contribution'.

[2] Data quality assessment, the subject of the SETAC workshop in Wintergreen, 5-9 october 1992, is the most important aid in guiding the sensitivity analysis in the field of empirical data. Sensitivity for choices of methods, e.g. for allocation, is the next part of the LCA sensitivity analysis. The sensitivity for the normative assumptions may show how near the alternatives are in a normative sense.

LCA information

As indicated above the life cycle analysis in the LCA instrument should preferably include as its final step, the formalized comparative evaluation. There is a choice here as to who makes this evaluation, a collective body or its representative, or private individuals or groups according to what they consider right. Each environmental group, for example, could then have its own set of environmental problem weights. In the latter case there are no "environmentally better alternatives" in a general sense, only differing opinions. There are two reasons to make this evaluation collectively instead of privately. First, the effects evaluated are truly collective; it is 'states of the world' that are at stake. That is the reason that environmental policy has become a public affair. The opinion of a group or an individual is therefore not a relevant basis for general evaluation. For this reason, the normative choice involved cannot be delegated to expert panels or other committees of technical specialists. Secondly, it is a heavy burden on individuals and organizations to make such an evaluation in each product choice themselves. The life cycle analysis should consequently include a set of publicly supplied weights for the environmental problems caused by a product system. Including such a set of weights in the standard for life cycle analysis does not imply that a private discussion on these weights should stop, or that this set remains the same for ever. On the contrary, the specification allows a public discussion on which environmental problem is now more threatening than another, each group trying through the political process to get official sanction for his priorities.

In all private choices the method for LCA that includes publicly decided weights, combined with further empirical information available to the decision-maker and not to government, could generate the right analysis. The macro instrument for environmental policy is the supply of a full LCA methodology, including a set of problem weights in the most practicable manner. Practicality could be enhanced greatly by two additions to the pure LCA method. First, the methodology may be supplied in the form of software that supports and also guides the analysis. The complexity of the analysis involved would be reduced greatly by such programs, which could be designed for specific applications. Moreover, the data requirements of all upstream and downstream processes are a main hindrance in executing private LCAs. Thus, in the second place, a database that gives the information on all main economic processes could solve the data bottleneck to a great extent. Process data on the environmental effects of production processes for most materials and waste handling are now largely lacking. It is practically impossible to generate that information[1] when making case studies.

Another practical question is what is the public body that is to supply the authoritative information on LCA methodology. The international nature of globally oriented LCA is reason to look at a higher level than that of the national state. Public bodies that could play a role at that supra-national level are regional bodies, such as the EC in Europe and the North American Free Trade Organization, semi-global bodies such as the OECD, and

[1] Currently, the European chemical industry has united to supply the anonymous data on all their processes involved in making bulk polymers. Their collective effort, now unique at a global level, may greatly reduce the current difficulties in making LCAs. The Plastic Waste Management Institute in Brussels is responsible for this project, see Matthews 1992. It is questionable, however, whether the widespread supply of such data can be a private affair only. Those expecting a comparative disadvantage would hardly come up with the data voluntarily.

fully global bodies such as UNEP, GATT and ISO. With the exception of GATT, all these bodies are now engaged in LCA development. The vital initiating role of SETAC, a private international body of environmental chemists and toxicologists, will become less important when such public bodies take over. Before the instrument is developed at this level, however, it is national governments and such private bodies as SETAC and PWMI (see the note above) that also play an important initiating role. An authoritative (as opposed to a scientific) role for private organisations such as SETAC and ISO will always be restricted to the level of a Code of Conduct, specifying how responsible practitioners should use the method. National governments might even go so far as to develop the standard LCA methodology themselves[1].

LCA incentive

The cultural incentive is the other face of the instrument. The assumption for the effective functioning of LCA methodology is that a significant number of people care about the environment. The motive might be strengthened by public policies, as through environmental education. Such long-term possibilities for normative change, of a more general nature than only LCA, are not the subject of analysis here[2]. Hence the incentive is assumed to exist privately, or some other normative instruments might be developed. Societal working mechanisms do not depend on everybody knowing the environmentally right choices and acting accordingly. If only a minority in the population is willing to generate the information and make some sacrifice to act according to it, broad effects may still be achieved through indirect mechanisms. A purely short-term-profit oriented firm, the most a-social variant of blind capitalism that can exist, might still engage in environmental product design. By such a design process the minority of environmentally concerned people might become its clientele, a highly desirable aim for such firms as well. Also, the failure to act in this responsible way could result market losses and in the part of the population that cares most about the environment, probably the better educated part, refusing to work with them, thus causing a rise in their production costs. More generally, having a bad name in whatever domain of public discussion, is highly undesirable for competitive firms that use their name or brand names in marketing.

Experiences in consumer reactions to environmental information indicate that a substantial part of the population is willing to sacrifice some effort and money for environmental reasons. Examples are the widespread support for separate waste collection and the market share for phosphate-free washing powders. The latter effect has been especially impressive as the information on the environmental advantages of phosphate-free washing have been plagued by contradictory and partial information. No full LCA has ever been applied here, let alone one based on a publicly standardized LCA method.

There is thus no compelling reason to engage in normative policies specifically directed at LCA. However, the general functioning of government is invariably value laden. The general legitimacy of government will make any public pronouncement on what is

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1 The Dutch central government methodology project on LCA has taken a firm step in that direction. It has supplied a practical manual, see Heijungs et al. 1992b.

2 One might doubt the legitimacy of the role of government as a normative educator. The independent citizen controlling government is then transformed into the dependent subject, whose thoughts are controlled. This is the case only if efficacy is assumed.

collectively decided to be good or bad into a normative statement. It always indicates a social norm and a good citizen should make his choices accordingly. Thus the purely informational standard methodology for life cycle analysis, by being a public standard, also gives normative support for making choices in what it defines as an environmentally more attractive direction. Thus no compelling reasons exist for separate normative elements in the macro LCA instrument. This stand does not support the conclusion that separate normative policies are undesirable and should not be developed. It is perfectly possible to support the normative status of LCA-based decisions by less macro instruments, as long as they are in line with the standard method. Many options are available, ranging from ecolabelling to showing how a well-known football star "always chooses the right product".

4.2.8 Conclusions

The standard methodology for life cycle analysis may be developed practically.

A general framework now exists and with minor modifications it can become the kernel of the instrument. All elements or components of that general framework have to be made operational, requiring the development of further LCA theory.

In the quantification in the inventory of the contributions of all processes required for the function of a product, the allocation of multiple processes is a major problem. That problem has been solved here in principle, in the assignment of social causation and physical causation to their respective places.

The specification of potential environmental effects, in the classification or impact analysis, requires further choices on which problems to include and how to include them in the analysis operationally, in terms of models. The position has been defended to exclude problems with a mainly local scale of occurrence, and to exclude local, site-specific effect analyses of problems from LCA, at least for the time being. A list of about sixteen problem items then results.

Some progress has already been made in making these operational, especially such global problems as climate change and ozone layer depletion. Recent developments at several research institutes indicate that other problems are on the verge of being made operational for LCA.

To be broadly applicable in choices of products in society, the method should include a formalized weighing procedure for the scores on the different problems in the environmental profile of a product. These weights should be supplied authoritatively to society, through a public decision.

Thus the information content of the society-environment interface may be made operational at the highest target level possible.

The working mechanism at the government-society interface is very limited. Its main activities are supplying the methodology, including the set of weights and supplying as supporting tools software programmes and a general data base on main economic processes.

Further normative policy elements, such as education, are not strictly required but might be supportive. Some normative stance is implied even in the provision of the methodology and tools alone. Also, the motivation for environmentally oriented choices on products seems quite widespread already.

No public application of the methodology to cases is required. However, if more specific product policies are developed, the methodology should be used by governments as well.

4.3 SUBSTANCE FLOW ANALYSIS AND SUBSTANCE DEPOSIT

4.3.1 Introduction

The second main instrument selected for macro environmental policy is the substance deposit. It has an economic mechanism at the government-society interface. The substance flow analysis is used primarily to make the society-environment interface operational. At that interface the deposit system is in a sense, the mirror of the life cycle analysis. Life cycle analysis ascends to the most aggregate level of the target, all substances for "total environment", and combines this with the highest level of process aggregation compatible with it, the "life cycle of a product". The substance flow analysis for the deposit system, at the same interface, chooses the most aggregate level of object, "all processes in society", and combines this with the highest level of target compatible with it, the "one substance for one problem level" (level d in table 2.3.3). When this interface becomes operational, and the signs are that it will, the target level might be extended to "groups of substances for one problem" (level f), or if the weighing of problems becomes operational, to "one substance for total environment", (level i). The substance flow analysis of individual substances remains the empirical basis for such further extensions. That analysis is the subject of the next section, 4.3.2. The economic government-society interface, added to the substance flow analysis, results in the substance deposit. That economic mechanism is the subject of section 4.3.3.

As with the life cycle analysis, the substance flow analysis as treated here is based on collective research at CML. The origins of the combination of substance flow analysis and substance deposit go back to a student research group in 1987, in which Udo de Haes and this author actively participated[1]. The first CML publications on the subject were Udo de Haes et al. (1988) on the substance flow analysis, and Huppes et al. (1987) on the substance deposit. Both interfaces, going under other terminology, were refined in the subsequent years, in connection with several projects for the EC, the Dutch central government, and a number of provinces, of which the province of South-Holland was the first to employ the more refined method of substance flow analysis.[2] The current position of the substance flow analysis is described in more detail in van der Voet et al. (1989), with references to the broader historical development of the method of substance flow analysis[3]. The substance deposit has been described most recently in Huppes et al. (1992). Its description here is more in terms of the current framework. The substance-flow-analysis-based substance deposit has ripened as an instrument much more than the standard methodology for life cycle analysis. Astonishingly, from the instrument application point of view, the practical use of LCA has proceeded much further, with active interest shown by many governments, international bodies, and private firms and organisations. The substance deposit has not been proposed at a governmental level yet,

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[1] Guinée, now a leading specialist on life cycle analysis at CML, was one of the students in that group.

[2] See van der Naald 1989.

[3] The materials balance approach has been developed since the Seventies, with Kneese et al. 1970 setting out the first general theoretical framework linked to economic analysis. There, all substances were treated under one general mass balance, hence the name. The separate treatment of individual substance dates later. Nijkamp 1979 still treats the materials balance principle at an aggregate level. A survey and theoretical analysis can also be found in Ayres 1989, with a breakdown oriented towards individual substances.

although several studies on it have been executed both for the Dutch central government and for the EC.

4.3.2 Substance flow analysis

The substance flow analysis (SFA) is a method to systematically organize the data on a substance. Its potential applications are much broader than only serving the substance deposit. The deposit scheme is limited to the flows of the substance through all material processes of the economy, while SFA also covers all environmental processes as well. The deposit scheme is restricted to the administratively defined boundary flows of the system; the SFA may also go into the depth of the system, analyzing sectors, groups of related processes, and individual processes in the economy. The deposit scheme is restricted to an administratively defined geographic level, while the SFA can effectively be applied at ecologically defined geographic such as the North Sea and the Rhine basin as well. The substance deposit is but one instrument for environmental policy, while the SFA can support many more. Some of these broader aspects will be touched upon. The main emphasis, however, is on the inflows and outflows of a substance through the material economy as the basis for building the society-environment interface of the substance deposit instrument.

Analysis of all flows of a substance

The crux of the substance flow analysis method is the integrated examination of all flows of a substance or group of substances within a geographic system. The mode of analysis is that of systems analysis, indicating inputs and transformations into outputs. The economy and the environment, two of the three subsystems that are distinguished, are subsystems that here are never merged into one. Such a fusion would prevent any analytic insight possible into what the economy does or can do to the environment. The two subprocesses of the total system thus defined, the economic processes and the environmental processes, may be viewed at any level of geographic aggregation. At the level chosen, the system can be described externally in terms of its inputs from, and outputs to its surroundings[1] and internally, in the first instance as the flows between the subsystems distinguished. The type of surroundings to be specified for the empirical analysis of substance flows of the economy-environment system is still quite broad. It encompasses the substrate, or lithosphere of the geographic unit analysed. Specifying the relations with the lithosphere is important for two reasons, first, depletion problems are related to the extraction of depletable resources from the lithosphere. Secondly, bringing substances into the substrate instead of the environment may help solve several pollution related environmental problems.
Thus the substance flow analysis is extended to also include the substrate. The overall system now consists of three main elements:

◇ the human economy, also called technosphere or antroposphere
◇ the biotic and a-biotic environment, also called the biosphere or ecosphere
◇ the substrate, also called the lithosphere or geological system.

[1] The systems-analytical term "environment", as anything outside the system defined, has been reserved for one subsystem here, the natural environment. Therefore I use the term "surroundings" to denote the systems term "environment". In Dutch and German, the term for the biotic and a-biotic environment is "milieu" and "Umwelt" respectively. In these languages, the confusion with general systems theoretical terminology does not occur.

The relations with what is now left of the surroundings are very limited; for substances they involve dust and larger objects coming from space, and gasses and space craft going into space. The inside of the earth, the environment of the substrate only, delivers several gases and energy. For energy flows the emphasis would be different of course. There, the main flows come in from the sun and from the inside of the earth and exit to outer space. The substrate, as a subsystem, differs from economy and the biotic environment in the much slower rate of geological processes as compared to economic and environmental processes several orders of magnitude slower. For the short-term analysis of policy instruments, the internal processes in the substrate may usually be disregarded. For some substances the processes in the substrate may be important in the general substance flow analysis but not for instrument analysis. Examples are the sulphur flows into the environment through volcanic eruptions and the natural leakage of CO_2, CH_4, and oil from underground reserves.

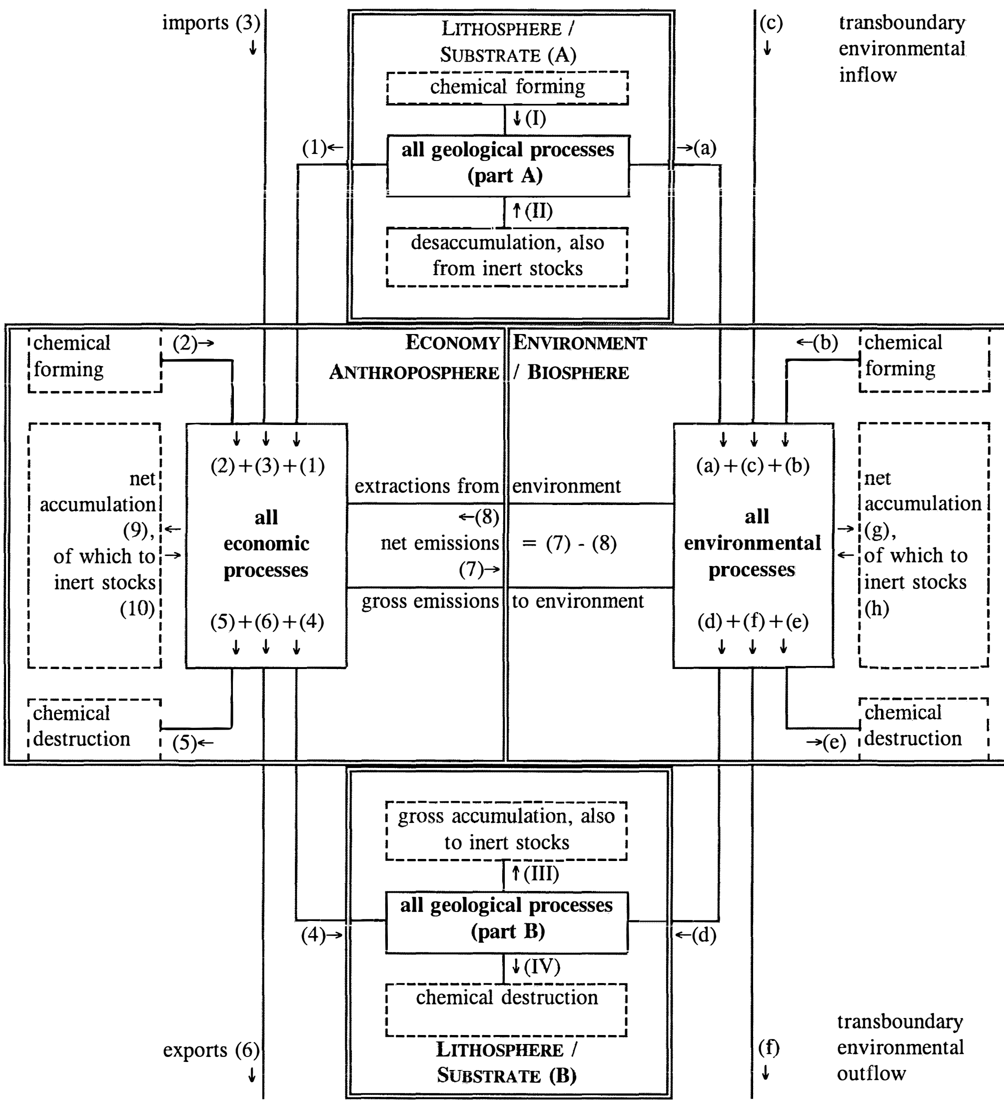
Systems relations often can more effectively be depicted graphically than described in words. For the further description I now turn to the most general substance flow scheme, see figure 4.3.1, after Udo de Haes et al. (1988). The three main systems elements are in double lines. Processes are in single lined boxes, flows are single lines with arrows, and some internal elements are in dotted boxes.

Flows with the surroundings
The system analyzed is the economy-environment-lithosphere system and its surroundings. These surroundings are comprised of other such systems on earth and in space. The flows with the latter are negligible or totally absent, except for energy flows. These flows have not been indicated in the scheme. Thus the first types of flows are these between geographical systems. There are the two inflows from other systems, imports from the economy of other geographical systems (3) and a transboundary environmental inflow (c) and the two similar outflows to other systems (6 and f). No cross flows are assumed to exist, e.g. from a foreign environment to the system's economy or from the economy to a foreign environment. If the geographic unit is the world as a whole, there are no flows between geographic units, and these four flows do not exist. All flows remaining are between or within the three subsystems of the geographical system analysed.

Flows between subsystems
With three subsystems distinguished there are six types of flows between them, grouped in three pairs. The first pair is between the economy and the lithosphere. The first flow is from the lithosphere into the economy (1), as the mining of ores and the second from the economy into the lithosphere (6), e.g. a given substance stored in disused coal mines and oil and gas domes or becoming stabilized and covered as occurs in certain waste dumps. The second pair is that between the environment and the lithosphere. There is a flow from the lithosphere into the environment (a), e.g. through weathering as a form of natural "mining", and there is a flow from the environment towards the lithosphere (f), e.g. when sediments become fully covered and stabilized.
These four flows from and to the lithosphere are directly relevant in the analysis of problems involving the depletion of geological resources.

The third pair of flows is that between the economy and the environment. Gross emissions (7) go from the economy to the environment, causing problems there. Less

FIGURE 4.3.1 SFA SUMMARIZED IN THE GENERAL SUBSTANCE FLOW SCHEME FOR ONE SUBSTANCE OR GROUP OF SUBSTANCES



obvious perhaps, many substances are extracted from the environment (7). Large flows go from the environment into the economy through the processes of agriculture, forestry, and fishing. An example also is the phosphates produced when making drinking water from surface water. The difference between gross emissions and extractions are net emissions. In exceptional cases, these net emissions may be negative, as used to be the case with phosphates and nitrates in agriculture. These are the two flows most directly related to the several environmental problems of pollution, and also to some depletion problems (e.g. drought and environmental undernutrification[1]).

Inflows and outflows within subsystems
Within each of the three subsystems an inflow may result from chemical forming[2] (2, I and b, in the economy, lithosphere and environment respectively), again mirrored by the outflows through chemical destruction (flows 6, IV, and e, respectively). Examples are the forming of CFCs in the economy and their destruction in economy and environment. At a global level these three inflows and three outflows sum up all inflows and outflows that exist. If substances are defined in terms of elements, there is no chemical formation and hardly any physical formation for most of them. For the elements the sum-total of the amounts in all subsystems is constant and flows may only influence the distribution between them.

Accumulation in subsystems
Finally, there is an accumulation category in each subsystem (9; III minus II; g). All flows are recorded for a period of time, e.g. a year. There is no special reason why the total inflow should be equal to total outflow. There then is a balancing item that indicates the increase or decrease in the total stocks in each subsystem that has occurred in that year, the net accumulation. As long as a unit of the substance is in the economy (or the environment or lithosphere) it may function in a process, this also applies to the amounts of the substance accumulating. Accumulation is not a flow or a process itself, it is the recording of a special kind of "net flow". There is a close parallel to the recording of financial flows of costs and proceeds. In bookkeeping terms, the accumulation is the balancing item of the accounts, indicating an increase or decrease in total wealth. The balancing item itself is not "proceeds" or "costs", nor is it an economic activity.

However, accumulation may not only be a balancing item in the abstract sense, as is the case with cadmium in nickel-cadmium batteries circulating in the economy. In some forms, part of the accumulation may be an accumulation in a more real sense, as becoming an inert stock[3] (10), e.g. a waste that is "kept" in the economy, e.g. certain effectively stored chemical and nuclear wastes.

Processes
Processes are the items that cause and change all flows of the substances. It is these processes that keep the flows of the substances going, by definition. In general systems theory, and in the general substance flow scheme, processes are defined in terms of their inputs and outputs. Thus, there is one process in the economy named "all economic processes". Below that general level of analysis more specific processes may be defined

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[1] Indirectly, these depletion problems may also cause a deterioration in other value areas, such as the quality of nature and the availability of other natural resources. The further modelling of such mechanisms is not part of SFA but of course closely related. See Huele and Kleijn on "life support system" in van der Loo et al. 1992.

[2] In principle forming is not limited to chemical processes. Elements may be formed through nuclear processes. For some rare elements, e.g. tritium, this may be a substantial inflow. Such physical forming is included in this category.

[3] The terms sources and sinks, often used in relation to the carbon dioxide problem, might be defined in the systems terminology developed here. Sources could denote the two types of inflow (from other subsystems plus chemical forming) plus the gross mobilisation from inert stocks. Sinks could denote the two types of outflow plus the increase in inert stocks in a subsystem.

that together comprise "the general process." In the environment the processes involved are physical-chemical processes (evaporation, deposition, leaching, sedimentation, sediment stabilization, decomposition, etc.) and biological processes (bio-accumulation, bio-degradation, etc.). In the lithosphere it is the much slower geological processes that cause flows. They are all physical-chemical.

In the economy, these more specific processes are tied up with production, consumption and waste treatment. Economic processes comprise any activity in society that has a material aspect, here in terms of the substance analyzed. That is one level of definition, the chemical-physical material level. At the same time these processes are economic entities in a symbolic, especially financial sense. In that respect they are not only governed by the laws of nature but also by human motives, knowledge, rules and institutions. The processes distinguished in life cycle analysis are described in terms of a *knowledge format*, comprising a name, values, functions, and material (chemical-physical) aspects. The minimum format for the general substance flow analysis covers only one part of the material description, the part on the substance in question, and only one name, "all economic processes", or more specific processes within that *economic metabolism*. When the flows are analyzed at a more concrete level, the subprocesses concerned also need a more concrete name. In many applications of the substance flow analysis a more detailed format for process data may be required. For the substance deposit, all inflows and all outflows must be linked to such specific processes. If economic mechanisms, in the sense of financial and broader social mechanisms are to be included in the analysis, the format should include functions and (financial) values as well. For the analysis of the interface of the substance deposit all such non-material aspects are not relevant, however; the processes involved with the substance must be named and specified only in terms of the substance[1].

Symmetry

The resulting figure is symmetrical along both the horizontal and the vertical central axes. This feature allows the aggregation and desaggregation of geographic units without any alteration in the terms of the schemes. The figure can be folded twice. First, it may be folded horizontally, putting substrate B on substrate A. Then inflows and outflows are reduced to "net inflows", for both the economy and the environment. When the diagram is folded a second time, i.e. vertically along the boundary line between economy and environment, the resulting scheme gives the net inflows and total accumulation of the combined economy-environment system. This symmetry may not be important in itself. Systems may be defined in any way as long as the boundaries are clear. Our experience has shown that once boundaries have been introduced based only on a subject studied, e.g. "the flow analysis of agriculture", that detailing all boundaries becomes an arbitrary activity with one researcher doing it one way and another researcher doing it another way. Practically speaking results then become incompatible. The symmetry ensures that any system is defined as a clearly delineated subsystem of a more encompassing total system.

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[1] For an analysis of indirect effects of measures the broader process characteristics of course are essential. When analysing cadmium measures, e.g., knowledge on the inelastic supply of metallic cadmium is indispensable.

What SFA is not

The general system pictured in figure 4.3.1 is limited to the quantified description of the flows and accumulations of a substance for the three subsystems. "The economy" is not analysed in terms of economic growth, GNP, development of economic sectors, or even cost assessment of emission-reducing measures. The scope of the basic model of the economy, on the left hand side, is restricted to flows of a substance or group of substances through the economy. On the right hand side, "the environment" is not analysed in terms of for example health risk assessment, forest death or endangered species: only the substance flows through the environment are given. However, the basic model can be extended in principle to include as many of such supplementary models as required. When the SFA method for scenario analysis is used, some assessment of the environmental impact of the substance flows is necessary to obtain a picture of the effectiveness of policy measures. On the other hand, at least a rough estimate of the costs of policy measures and of their allocation is needed to assess efficiency, and determine who is to foot the bill. These additional models, simple or complex, are not the issue here. The question here is how the substance flow analysis may define, or help define, the society-environment interface of instruments in the desired manner.

The interface of society with the environment and the lithosphere

The interface of society or, more specifically, economy with the environment and the lithosphere consists of an object part and a target part. The object is all economic processes in the administrative unit to which the instrument is to apply. This object is the kernel of the economy part of SFA, with the geographic level adjusted to the desired administrative level. The highest level, the global level, is the simplest in its boundary relations. The smaller the administrative unit, the greater the imports and exports will be as a fraction of total inflow and total outflow respectively, and the larger will be the number of different import and export items containing the substance. At the global level, the only inflows into the economy are those from the substrate and through chemical forming, and through extraction from the environment and the only outflows are those to the substrate and through chemical destruction, and to the environment. For instrument design for the substance deposit, a vital distinction should be made between all inflows directly related to the environment and those not related. In the long term, there is an equality between on the one hand the net inflow from the lithosphere (1 minus 4) plus net chemical forming (2 minus 5) and net emissions (7 minus 8). This long-term equivalence forms the basis for the deposit instrument. Strictly taken the equation has to include net accumulation as well:

> *net extraction from the lithosphere plus net chemical forming minus net accumulation*
> *equals net emissions.*

Also including imports and exports the full equation becomes:

$$[(1) + (2) + (3)] - [(4) + (5) + (6)] - (9) = (7) - (8)$$

The mainly abstract "accountancy" nature of this net accumulation does not allow its inclusion in a real global instrument. The society-environment interface for the substance deposit is now defined as the left half of the equation, excluding accumulation:
The interface of the substance deposit system has a twin object:

◇ for deposit payment it consists of the flows of the substance into the economy through extraction from the lithosphere, chemical forming and imports *[(1) + (2) + (3)]*

◇ for refunding it consists of the flows out of the economy to the lithosphere, through chemical destruction and exports *[(4) + (5) + (6)]*.

In addition, one could make the real flows to "inert stocks" the basis for repayment. By definition, emissions from inert stock can occur only if the substance has been taken from inert stocks. Taking the substance from inert stocks into active economic processes would then be a basis for deposit payment. This option will not now be worked out here. One problem is how to define the boundary between stocks in economic processes and inert stocks in the economy.

Aggregating the substance target

The target level in the substance flow analysis is primarily the lowest level of aggregation possible, i.e. one substance (level a in figure 2.3.3). At the level where the analysis is made the data on the processes fit into the scheme. The overall inflows, outflows and accumulations are given, as are the extractions from the environment and gross emissions to the environment. There is no relation yet to higher level targets in the environment, such as one problem, several problems, or "the total environment". Remaining at the lowest level of target aggregation is not desirable when developing macro instruments. The normative meaning of the emissions would have to be assessed in any case anew. Also, basic discussions would have to be repeated for each substance. When the effect analysis of an individual substance is embarked on, myopia could obstruct a reasonable instrument choice. Scientists can always explain that the administrative level is not relevant for the environmental analysis and evaluation. Better knowledge of effects requires ever more knowledge about ever smaller geographic units. However true this might be, that approach precludes the development of macro instruments for environmental policy. The strategic discussion on macro instruments is intended to end in a more aggregate target level for the interface.

How could the more aggregate target be formed? The task of aggregation is nearly the same as that of setting up the classification in LCA. There the need for a higher level analysis was more pressing because the sheer amount of interference information was overwhelming and could not support an evaluation of alternatives. A separate classification and worked-out evaluation system were therefore introduced in LCA. With the deposit instrument, the situation is somewhat different. The deposit level (and also that of the emission tax) should indicate the damages expected to be caused by the substance. Neither empirical models nor evaluation methods are practically available, however. At present a separate line of reasoning for each substance decides the deposit level. For the three hundred or so substances currently regulated, this setting of deposit levels would be an administratively chaotic and publicly incomprehensible activity, as was the evaluation of product alternatives based only on the interference table. Could the same vein of reasoning applied in the LCA instrument also simplify the situation for the valuation of the substance flows? This would seem to be the case. The basic choice that led to a solution in LCA was to funnel the analysis to supra-local problems related to interferences through a globally uniform effect chain. The resulting problems were the input into a further evaluation procedure. If only the step to the problem level could be made, the number of discussions on tax levels could be reduced from several hundred to a dozen or two. At the same time, the discussion would be simplified since it would solve the different parts separately.

Establishing the potential global warming contributions of all relevant substances, for example, is an extremely complicated affair already. How long should the time horizon in the models be (one hundred years (better: indefinitely; are two dimensional models really good enough for this task (no, they are not); should indirect effects be taken into account (yes, they should). Such discussions about problems are now independent. If these discussions were mixed with those on the costs of technical alternatives, on the indirect contributions of different problems to each other, and on the relative and absolute importance of all environmental problems, they are unlikely to lead to a well-reasoned level of the substance deposit or emission tax.

Setting the target level at the level of "all substances that contribute to one problem" (level g in figure 2.3.3) thus has clear advantages when implementing the economic instrument for substance policy. That advantage is bought at a certain expense. The aggregate instrument cannot be differentiated in terms of differences at a lower scale level that really are, or will be present. For all non-global problems it makes some difference where the environmental interference took place. Even for the global problem of climate change the contribution made may depend somewhat on the location and even the season and even time of emission. The strategic choice of macro instruments has already been taken with these disadvantages in mind. If a tax level has been defined for several problems, a set of weights on problems implicitly is given, translating these problems into "the total (supralocal) environmental problem".

Establishing the problem target would require the renaming of the substance deposit as a problem deposit. I retain the term substance deposit, because the name is familiar and because the substance flow analysis continues to play a central role in its set-up and implementation.

4.3.3 The economic government-society interface: the substance deposit

The substance deposit has been chosen in principle as the most aggregate economic instrument. With its society-environment interface established in more detail above, the task here is to work out its administrative implementation and indicate its functioning. The latter aim will be achieved by a comparison with similar emission taxes.

Administrative aspects

The administrative activities required are the deposit payment on the inflow of the substance into the economy and the refunding at outflow. For many substances, especially the durable types, the number of different inflows is usually limited. For hazardous elements, such as heavy metals, inflow is restricted to extraction from the substrate through mining. Man-made chemicals are generally produced by a limited number of larger installations. Imports will pose the most serious implementation problems, especially if the administrative level is low. At the global level, however, that problem vanishes as there are no imports and exports there. The refund, by nature, is easier in administrative terms than the deposit payment. There, regulatees have a clear incentive to specify what they destroy, put back into the substrate, or export. In many instances the main administrative burden will be border controls. The inflow of the substance in small and varying amounts in the form of many different products is causing the trouble. At a high administrative level, the implementation of deposit payment will often be relatively easy compared to the emission measurement of individual installations.

The requirements for the administrative implementation of the system are very much comparable to those of excise taxes, such as duties on alcohol for human consumption and gasoline for cars. Taxing at inflow is fully equivalent to the deposit payment at inflow. Repayment at export and denaturation are very similar to the refunding at export and chemical destruction. Alcohol "back to substrate" is not a basis for repayment. With alcohol one pays for the potential human consumption, with the substance deposit one pays for the potential emission.

Any less aggregate grouping of processes than that encompassing the total economy creates a problem because then the total economy is divided into parts that require their own boundaries. From an environmental point of view these boundaries are arbitrary; any emission is an emission[1]. From the private firm point of view, however, it can be profitable to bend the boundaries a bit. The ideal is to acquire the substance without deposit payment and than sell it outside the deposit domain and have the deposit "re"-funded. Any extra boundary requires extra administrative problems and creates extra opportunities for fraud as well as an incentive for it[2].

The existing national duty and excise offices could have a major role in implementing the deposit system. The administrative activities required seem limited and simple compared to those required for current direct control instruments.

Societal working mechanisms
The functioning of the substance deposit is approached by, first, a short discussion some general notions on deposits. Its societal functioning is indicated next, starting at the micro level and moving to the macro-level in a number of steps.

As with all macro instruments, the processes influenced will not be the same as those addressed by the instrument. The societal working mechanism is primarily economic. Within the administrative unit, the substance will be assigned a price, and if priced already, the price will be raised. Losing the substance to the environment will therefore become more of a loss. All decisions taken within budget constraints, i.e. a broader category than decisions taken in a market context, will adjust to the new price levels. On the one hand activities and their products associated with high losses to the environment will become relatively more expensive, with a consequential substitution to other activities

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[1] This also holds for the distinction between public and private ownership, e.g. of waste processing installations. Public installations should be subject to exactly the same economic instruments, i.e. they should be fully "within the system". The authority implementing the deposit system, or any other instrument of environmental policy, should not be the owner of the public installations.

[2] A proposal has been developed in the Netherlands, by CLM-Utrecht (not CML), to tax farmers on the basis of the bookkeeping of all nutrients. This would require both a check on all flows between farms, from and to other firms in the production column, and on the transformations presented at farms. The initial check may be administrative, on overall systems consistency. A further check, on truth, is more difficult. Having alcohol excises implemented at the level of retailers would pose similar problems.

and products. On the other hand, there will be an incentive to change technologies to prevent emissions. These general mechanisms are the same as those generated by an general emission tax at the same level.

Certain ideas on the functioning of deposits would set them apart from emission taxes. One such idea is that a deposit, as against a tax, need not cost money to those paying the deposit, since they can return the item if they wish to. Return the object for which the deposit has been paid and no expenses are involved (apart from financing costs and the real costs of having the object returned). Thus, lower income groups would be hurt less by a deposit than by a tax. The deposit on drink containers in many countries is a common example. However, this idea is misleading and wrong. Imagine that instead of the deposit, there is a tax on throwing away drink containers. As with the deposit, the tax can be completely avoided by bringing back all empty containers, for example. The throw-away tax would have the same effect as the deposit, on all behavioral choices, and also on the real costs resulting.
Secondly, in most instances the substance deposit would not translate into a product deposit. Nor would the emission tax. They both would result in changing relative prices, in changing technologies, and in lowering the market volume of the consumptive activities concerned, in return for environmental improvements. If both are applicable, they both might generate market-based deposit systems, in order to prevent the loss of the deposit paid or to prevent the tax being levied. Both would result in proceeds for the implementing agency, thus replacing other taxes or financing extra public outlays. See figure 4.3.2 for the near equivalence of substance deposit and emission tax.

At the micro level, the functioning of a substance deposit and emission tax could be very similar. However, many processes have their emissions at irregular intervals, at many different locations, or even in diffuse forms. Measurement of emissions is then difficult or impossible. Measurement would always be an individual affair that involves checking on each specific process. For each such process the deposit might be used as an alternative, at the micro level. A factory processing cadmium into dyes will emit cadmium in diffuse and inhomogeneous waste flows that may be difficult to monitor quantitatively. However, cadmium going into the plant can be measured quite precisely, as is the case with cadmium coming out of the plant. Thus, a cadmium deposit imposed at the level of the plant could in fact tax all diffuse cadmium emissions much more easily than a direct emission tax itself. Similarly, the nutrient losses of a farmer, e.g. phosphates, are highly diffuse, taking the form of several complex biological and physical processes. What can be measured is the amount of phosphate going into the farm and the amount leaving the farm as produce. A deposit could then be introduced to tax the losses wherever they may occur on the farm. These very real differences relate only to differences in implementation, however, not to differences between the two instruments in societal functioning.

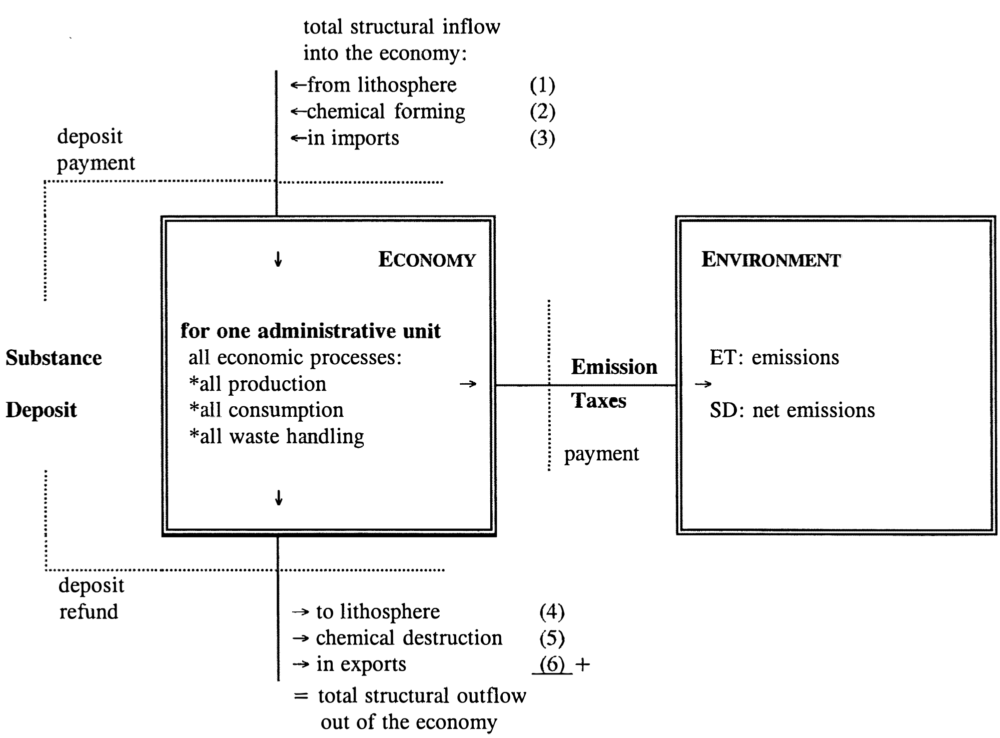
At this micro level the problem arises of handling flows in outputs that are processed by others, e.g. wastes. These do not constitute an emission and are not taxed. They leave the firm without being emitted and are thus eligible for a refund. Without proper handling, however, the waste may become an emission. Hence the deposit system, and the emission tax, might be extended to each directly related process that causes emissions. The farmer would have his deposit returned for the phosphate in manure delivered to a manure processor. That processor would have to pay the deposit again. He could obtain a refund for any product coming out of his process. Implicitly he also would pay for any emissions. If such a deposit system on the substance were introduced for all processes, it would cover all emissions in society. It would be equivalent to an emission tax of the same amount per unit of the substance on the same processes. Such a deposit system would not be a macro instrument however, as it would apply to individual processes only, as does the emission tax.

As indicated, the deposit system can be transformed into a more macro type instrument by grouping processes, as the objects of the instrument, into larger units. Possibilities for grouping are given in table 2.3.3 in ascending order, starting at basic installations. Some of these steps will now discussed to indicate the working of the deposit instrument. First, let us assume that one farmer and one waste processor are together subject to the system. The farmer would pay the deposit, while the waste processor would receive the refund. The waste processor will now be willing to pay the farmer for his formerly negatively valued wastes, if the deposit is high enough. The reason for this change is that the proceeds of waste processing now include the refund for phosphates worked into its products. The waste processor will not pay the farmer the deposit as a separate sum, he will raise the price he is willing to pay for the waste. The farmer, who has to pay a deposit for any phosphate entering his farm, will now think twice before throwing away phosphate containing wastes. He could recycle these on his own farm, replacing some inflow of phosphates-with-deposits, or he could now sell the waste to the waste processor, if the price is more attractive than net reuse proceeds at his farm. His supply function of the waste product now depends on the technologies for reuse becoming profitable because of the deposit payment. The combined system of the two firms thus leaves out the administrative activities of refunding and payment. It leaves these intermediary payments to the market mechanism, stimulating any technology that reduces emissions. There is still no difference to the functioning of the equivalent emission taxes.

Next, all farms might be taken as a whole, including the upstream and downstream processes which together constitute the agricultural production column. Any amount of the substance going into the production column requires the payment of the deposit. Any amount leaving it gives right to a refund of the deposit. Emissions within the production column are thus effectively taxed. How would the processor of agricultural waste operate in this situation? He would no longer receive a refund for his products, that are mainly used as agricultural inputs again. He still has to pay the farmer for acquiring the waste product to be processed, otherwise the farmer would not deliver his waste product. Would his activities stop? No. The incentive for his production is raised by the amount that his products can avoid the deposit payment on virgin inputs into the agricultural production column. Thus in the whole production column, all activities are geared to a more efficient use of the deposited input and, finally, to a smaller loss of it to the environment. The deposit on phosphate would be refunded for produce sold outside the sector, e.g. to consumers. These would then have no incentive to change to lower phosphate products. Nor would there be an incentive to prevent phosphate emissions from household wastes. An emission tax on all agricultural processes, and not on consumers would have the same effects.

The ultimate grouping is "all processes" (level 13), which now also includes all other sectors, such as those producing phosphate-containing washing powders, and all households and all household waste processing. As the substance flow analysis indicates, there are only three types of inflow at that level, extraction from substrate, chemical forming, and import. There are also three parallel/analogous types of outflow. See figure 4.3.2 for the outline of the deposit system compared to the equivalent number of emission taxes on all processes emitting the substance. All black lines are substance flows. The dotted lines are payment barriers, either for the deposit system or for the taxing system. At this general level of analysis the now macro deposit instrument would function very much as an emissions tax on all processes involved. Of course, there would now be a very substantial difference in administrative implementation.

FIGURE 4.3.2 SUBSTANCE DEPOSIT AS A NET EMISSION TAX THROUGH INDIRECT EMISSION MEASUREMENT



Differences with emission taxes

Having indicated all similarities in functioning between the deposit system and emission taxes I will now turn to a number of differences with the emission tax. A main difference is related to taxing gross or net emissions. An emission tax taxes gross emissions, that is flow (7) in figure 4.3.1. The substance deposit taxes net emissions, that is flow (7) minus flow (8). Extracting an amount of a substance from the environment is a "negative emission". Under the deposit scheme, activities aimed at this result receive an implicit subsidy for their positive external effect. The subsidy is not paid directly. It is the "free entry" of the flows from the environment into the economy that results in the subsidy. Making the product chemically, extracting it from substrate or importing it from other countries would all have led to deposit payment. Extracting it from the environment does not. At the same time, the chemical destruction of the unit of the substance that has come into the economy this way would result in a refund.

A second difference is that the problem of missing markets, the case in much waste handling, is handled better by the substance deposit. The operation of both substance deposit and emission tax depends on the market signals given through price changes in all related markets. If there is no market or if the information content in the market is very low, as usually is the case with the processing of mixed wastes, the price signal stops. There is then no longer any effect at "the other side of the market". The deposit, however, applies from two sides. From the inflow side it works downstream, towards final waste processing. From the final outflow, with refunding, it works upstream towards the inflow point. Thus, the deposit scheme can still function when a market is lacking, while the influence of the emission tax stops at that boundary.

A third difference is related to the timing of payment. The processes leading from inflow to outflow may last for several years. The use period of products may extend for decades and, in exceptional cases, for hundreds of years. With the emission tax, all activities that delay the emission are attractive for financial reasons. Accumulation in the economy, as through working hazardous substances into building materials or processing cadmium into nickel-cadmium batteries are attractive. On-site "waste-to-be-processed", now already a well known phenomenon in the Netherlands to delay and eventually avoid the real costs of waste processing[1], will become still more attractive with an emission tax. With the deposit system, no mortgage on the future is possible; the contrary is true. The money for potential future emissions has to be borrowed, since the deposit is paid as soon as the substance enters the economy. The financing costs in long-lasting applications of a substance are not a virtue of the system though. Particularly long-lasting applications where a certain return is expected, e.g. large emergency batteries in hospitals, will become a financial burden under the deposit system. With excise taxes, such as those mentioned on alcohol, the taxing authorities have worked out methods to at least partially prevent undue costs of financing. However, the financing costs may also be seen as deterrent to shifting problems to the future, in which case they are justified[2], at least to that extent.

Finally, there is the difference in administrative applicability, as opposed to societal functioning. It is clear that both instruments differ, fourthly, in their domain of application. In some instances the deposit system may have clear advantages, i.e. where practical emission measurement is not possible. The converse is also true. A substance is brought into the system in many small amounts in different imported consumer goods.

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[1] One of the largest single items of cadmium accumulation is the on-site storage of the waste from zinc production at BUDELCO in the Netherlands. This company, with SHELL as a main owner, has delayed the processing of nearly all its wastes for decades and now wants heavy subsidies on processing. The jarosite, a waste containing cadmium and other metals, is stored in large basins.

[2] The level of the financing costs depends on the market interest rate. There is no automatic relation between the level of capital costs and a justified level of deterrence to shifting problems to the future.

These, after their use, are burned in waste incinerating installations. The administratively easiest option may be to tax the incinerator to limit the emissions of the substance. This solution has no influence on the composition of the products causing the emissions, owing to the missing link in the market for waste. The extra implementation costs of the deposit system could be justified by the lower costs of prevention as compared to the emission tax, especially in the long run. The differences in applicability, of course, result in real differences in societal functioning.

International aspects of the substance deposit
If the deposit system is installed not in one country as the administrative unit but in several, these together could function as the geographical unit for the deposit system. There would then be no deposit payment on imports from countries within that area and no refunding on exports to countries within the area. The nett transboundary flows within the area would not necessarily be zero; some countries would receive more deposit payments than they would refund, other countries would make a loss. If this occurs in a political unity like the EC, clearing could be avoided, the nett effects being taken into account in the more encompassing distribution of funds taking place anyway[1]. When such a more general supra-structure does not exist, a clearing procedure might be part of the international 'deposit agreement'. If the whole world would join, the deposit payment would apply only to extraction from substrate and chemical forming, refunding would apply only to returning to substrate and to chemical destruction. All border controls for deposit payment and refunding could then be abolished.

If the system is introduced in one country, or a group, this might at first sight seem to go against GATT regulations that forbid financial restrictions on imports. However, this would not be a correct position. The system consists of deposit and refund together, as a unity. Thus an import that is re-exported fully does receive back a full refund on the deposit and no external effect occurs. If the substance is emitted no refund takes place of the deposit paid at the border. Is that an unallowable import tax? No, it is an emission tax, levied at all emissions there are, originating from abroad or not. See in this sense the contributions of the international law specialist Sevenster in van der Voet et al. (1989b).

4.3.4 Conclusions
The substance deposit on problems has been developed to maturity. It has never been applied in practice. This policy instrument applies to the totality of all processes in society. A deposit is paid on all inflows of a substance into the economy of society. There are three basic types of inflow: extraction from substrate, chemical forming, and imports from other economies. The deposit is refunded at outflow. The three outflow types mirror the inflow. They are, back to substrate, chemical destruction, and exports.

The society-environment interface is based on the substance flow analysis (SFA). This systems analysis of substance flows can be aggregated at the level of the administrative unit that implements the substance deposit. SFA can be applied for more applications than the deposit system alone. The environmental part of the substance flow analysis is not

[1] That is a position very similar to the one that income distributional effects should not be taken into account because other instruments exist to correct any unwanted effects.

aggregated enough for the macro instrument desired, it refers to flows of individual substances only. To allow a structured discussion on the level of the deposit, the target of the instrument preferably is expressed in terms of problems. As in the life cycle analysis, these problems can only be supra-local problems. Differentiation between areas within the administrative unit would lead to extreme complications in implementation.

The substance deposit functions similar to emission taxes on all processes involved. There are four main differences with the emission tax.
First, it taxes net emissions only, contrary to the emission tax that taxes gross emissions.
Secondly, the economic functioning of the deposit system is more immune to malfunctioning or the lack of markets than an emission tax applied to the same flows.
Thirdly, the deposit system puts a financing burden on delayed emissions, while the emission tax gives an incentive for delay.
Fourthly and most important, in many situations where the emission tax cannot be implemented because of problems with emission measurement, the substance deposit might be applicable.

Application at a supranational level is possible and has advantages in diminished border controls. Application at a national level is possible without being at variance with, e.g., GATT rules. The deposit is a method for taxing emissions within the country only, an internationally fully allowable activity.

4.4 CONCLUSIONS ON THE DETAILED DESIGN OF MACRO INSTRUMENTS

Before any conclusions can be drawn on the macro instruments designed, it should be determined whether the right instruments have been developed. With hindsight, other options might have become more attractive. One option is to replace the rather mild cultural mechanism attached to the life cycle analysis with a, much stronger, economic instrument mechanism, such as the deposit or a tax system. The economy environment trade-off in decisions could then shift substantially to the advantage of the environment. The same life cycle analysis would then be required for the society-environment interface, to discriminate quantitatively between good and bad (or bad and worse), as a basis for the financial incentive. The deposit system is hardly applicable since the object is no more than an analytically discerned group of processes required for the functioning of the product. The best option might be to tax products, as the objects nearest to the functional unit. The tax on a product could be based on its contribution to several problems, with a price set on each unit of problem contribution. In that case no comparative evaluation of alternatives is any longer required; that evaluation is accommodated in the tax structure and the tax levels. The tax could be levied on the overall environmental index. Only final products should thus be taxed, to avoid double taxation. That tax could be developed analogously to the value added tax.

Such a *life cycle tax* would seem to have the same results as a full emission tax scheme of the same amount for all substances concerned, their target being the same. There would be one major difference however. With substance deposit and emission tax the objects have a direct link with the target. With the life cycle tax, however, the objects are used as estimators for the target; they are not directly linked to it. Thus the life cycle tax is an *estimated emissions/problems tax*, one of its many possible variants. The emission tax would cover all factual variations in behaviour, while the life cycle tax would be based on historical averages. Like any other estimated emissions tax, the life cycle tax would invariably omit some relevant behavioral choices. In the operation of a chemical plant, the amounts of most emissions depend heavily on care taken during operations and maintenance. Differences between plants cannot usually be reflected in LCA. Materials for a product bought in a homogeneous market are taken into LCA only as averages of the production techniques used. Being better then the others, environmentally, may help improve the average in the long run. It does not give an edge on competitors. The emissions tax, by contrast, would reward any environmental process improvement in material production. Thus the similarity between emission tax and LCA tax is only very partial. The LCA tax can hardly cover any of the dynamic aspects of production and consumption.

There are also some practical problems connected to the life cycle tax. Effectively, emissions all over the world are taxed in this way, since many products are related to production processes all over the world. It is quite unusual to tax events abroad, outside a given government's jurisdiction. It might even go against specific GATT rules since such a tax would discriminate against the producer of the parts, materials, or products in question. There is a second practical problem in the LCA tax. It is always possible in LCA to differentiate subgroup averages in the averaged population of processes, a procedure that leads to slightly different analysis results. Processes may differentiate between steel and stainless steel for example as well as between different types and grades of stainless steel. For tax purposes the life cycle analysis would therefore not seem inherently precise enough. Changing to the actual processes historically used for the specific specimen of the product requires tedious data collection. How is it possible to specify from which plant common grid electricity came? Even if these data were specifiable, such an analysis would not make sense since in a homogenous market, the product share not taken by one producer is by definition, taken by another.

The conclusion to be drawn about the life cycle tax is that the tax would not have the dynamic advantages of an emission or problem tax or deposit system which it would compete with, or overlap. Furthermore, life cycle analysis does not lead to hard results. The life cycle tax might also be at variance with GATT rules. Moreover, it cannot form a firm enough basis for environmental taxation because its analysis component is too crude. It should be borne in mind, however, that this limitation is relative. Compared with environmentally-based differentiation in real value added tax, as proposed by many environmentalists (see for example von Weizsäcker 1992), the life cycle tax is of course a highly refined jewel.

The two macro instruments selected for more comprehensive design, the standard methodology for life cycle analysis and the substance deposit for environmental problems, both appear to be practicable in principle. The deposit system is more suitable for immediate use than the standard methodology for life cycle analysis. A standard methodology can be established for the latter, the rudimentary methods now available will have to be worked out further.

Both macro instruments can be developed on the macro level only at the expense of differentiation between situations that might in fact be different. The life cycle analysis would break down because of the complexities resulting from any "near to real" analysis. The deposit system would break down administratively, since the smaller and smaller geographic areas would result in immense boundary problems. Thus, for both instruments, problems should be defined precisely but in a generalised, non-local and non-site-specific way, relating all relevant interferences to each problem in an explicit, but generalised model. Problems, since they are modelled environmental effects, can be completely the same for both instruments.

The life cycle analysis has an empirical and a normative part. The empirical part results in the quantified contribution to a number of supra-local problems, specified with preferably globally uniform models which give the environmental profiles of the products to be compared. The profile can be normalized to indicate relative levels of the quantitative contribution to each problem. The normative part requires an explicit set of evaluative weights for the problems included in the analysis. Without it, a comparative evaluation between the products compared will generally fail to materialise, apart from the mostly trivial situation of a single alternative being dominant.

The substance deposits may be set at a level to reflect their contribution to several environmental problems. A deposit/tax level set for different problems, indicates the importance attached to these problems. Thus, a set of weights on problems is defined implicitly in setting the level of these taxes. Conceptually, the deposit system might do without explicit environmental problems, by setting a level of the tax for each substance separately. However, without introducing the problem level in the analysis, it would be next to impossible to set a reasoned level of the tax for all the different substances occurring.

For filling in the flexible response strategy the standard life cycle analysis has a very broad but weak influence, while the deposit system can be much stronger but covers a much more limited number of problems and substances. Both instruments may thus have a legitimate place together in the same strategy. They might overlap however in which case a selection of product alternatives might first be stimulated by the LCA analysis and, on top of that, by the financially stimulating effects of the tax. An unbalanced choice would result with different trade-offs in different choices, going against both the equality and the efficiency principle. The first solution would be to give prevalence to the strongest and most regulative instrument, which is financial. The substances on which a tax or deposit is effectively levied should then no longer be counted in the LCA inventory. This refinement has not been included in the LCA instrument as described. The second solution would be to increase the domain of application of the financial instruments. With financial instruments becoming applied broadly, the life cycle analysis would lose its place in the strategy and might be abandoned. This will not be the case for a long time to come, however.

For the coming decades, it would be nothing less than an achievement if both instruments were developed and applied effectively on a broad international scale.

PART 5 CASES
APPLICATION OF THE FLEXIBLE RESPONSE STRATEGY IN FOUR CASES

5.1 INTRODUCTION TO THE CASES

The aim in the analysis of the following case studies is not primarily the solution of specific environmental problems. The case studies are steps in the specification of the flexible response strategy for environmental policy. The aim is to test the applicability and describe the functioning of the flexible response strategy. The two most macro types of instruments that have been worked out here, the standard methodology for life cycle analysis and the substance deposit, play a central role. The two instruments differ fundamentally in both interfaces. At the government-society interface, they use a cultural and an economic mechanism respectively. At the society-environment interface, the differences are also pronounced. The life cycle analysis takes the economy as a starting point and relates the central element of the economy, the functioning of one product system, to effects on all problems in the environment. The substance flow analysis starts the other way round. It takes one environmental problem as its starting point, chooses one of the several substances causing it, and relates the flows of this substance to all societal, i.e. economic, processes.

The flexible response strategy was developed in the preceding sections as a solution to a quite complicated mix of problems in policy development. When reading the cases it is a good idea to bear this general analysis in mind. The main lines of reasoning here are summed up here. Environmental policy currently is at a crossroads. Population growth and the growing material consumption per head, subject to a certain local differentiation, severely increase the pressure on our planet in general. The last two decades, environmental policy has primarily been developed as "much more of the same". The set-up of instruments has remained basically the same during the last century and a half, since the flowering of the Industrial Revolution. In this longer period, environmental policy was concerned with minor corrections to activities, from a global point of view. Point concentrations with clear local effects have been administered quite effectively through either dilution being used to deal with pollution or the controlled concentration, storage or destruction of substances. Some processes are regulated in terms of their functioning and others have been forbidden altogether. Wastes have been controlled in dumps and in waste incineration installations. The production and application of certain substances have been forbidden. Given the continuing success of the industrial revolution these measures, normally implemented through direct regulations and public provision, are not enough. A more fundamental influence on production and consumption is required to prevent problems instead of partially solving, partly displacing them.

Prevention is the new theme, related to both process technology and the products consumed. The position taken here is that prevention might be realized with the familiar existing instruments, but only at great expense. The web of effectively enforced regulations would become so thick that there is a real danger of society becoming ossified. This would mean that society is unable to react adequately to new challenges, environmental and otherwise. Reaction time and the reaction energy required increase exponentially as the numbers of regulations and related procedures increase. Not only may the environmental problems remain unsolved for a given case, the general technological development will also initially not be influenced environmentally and will then grind to a halt. Growth of population and of production based on ever heavier investment in current technologies will certainly lead to environmental disaster. Doing nothing, disguised perhaps in the form of vague long-term declarations or ineffective measures, is the other path to certain disaster. The only way out of this dilemma is for private decision-making in technology, production and consumption to be more indirectly guided in environmentally friendlier direction by a "less visible" or even "invisible" hand. Influencing decentralized private decision-making with such an all pervasive invisible hand is the design aim of this study. If such a policy change were effected, it would be a fundamental change. It would bring environmental regulation into line with the societal steering mechanism introduced with the liberal revolution in the Renaissance. That type of societal steering marked the beginning of the industrial revolution. The same successful steering mechanisms may also help to solve the environmental problems it caused.

The hand may be invisible for those making decisions with environmental consequences. It can start working, however, only by active policy choices of types other than "more of what we had". The main choice proposed here is one for the *flexible response strategy*. In that strategy, solutions to problems are sought by systematically applying those instruments that treat the subjects regulated equally and leave them as much freedom of action as possible but that still are effective environmentally, much more so than current policies. The general nature of such instruments is that work on all relevant micro events but indirectly only, applying to higher, more macro, system levels.

The purpose of the case studies is to illustrate the applicability and the functioning of the strategy and the individual instruments in it. The first case study is on the standard methodology of LCA. It describes some partial working mechanisms of LCA, in the marketing and product design of milk packaging. The life cycle analysis in this strategy is thus *not* intended to work out the policy for a specific product. The case study on milk packaging consequently does not end with prescriptions of how governments should act with respect to milk packaging. It rather indicates that they would do better not to act at all. The use of LCA for the direct regulation of individual products is a last choice option here, only relevant perhaps if all macro instruments combined cannot contribute enough to increasing environmental quality.

The case studies on substances are by nature already more at the macro-level, connecting several types of economic processes. It is all the activities relating to the substance investigated that are taken into account. The substance is defined in relation to one main environmental problem. Toxic pollution is the environmental problem central in the second case, on cadmium. In this one case on a toxic substance cadmium represents all the heavy metals, all of which have similar working mechanisms, or it may even be seen as a representative of the more heterogenic group of all toxic substances, including those that are man-made and degradable. If a substance contributes to more than one environmental problem, it may be the subject of policy development twice. If substance deposit and emission taxes are broadly applied, such a substance would justly be taxed several times. It would be advisable, of course, to develop the policies for such a substance in relation to one another. At this stage of scarce information and few financial instruments there is not yet overlap. This is why in the third case, nitrogen and phosphorus, these two substances which are both central to the two problems of acidification and eutrophication, are treated together.

The fourth case is initially dual problem-oriented. It takes as a starting point current discussions on environmental energy taxes. The environmental arguments for energy taxes relate primarily to the problems of energy depletion and global warming. The energy depletion problem is analyzed separately, in Appendix 1 at the end of this section. It is reasonable to suppose that there is in fact no such problem. Policy development for a non-problem or at best a low priority problem makes no sense. The global warming problem, however, is here to stay. Assuming that the CFCs and the most serious (H)CFCs are phased out, it is primarily emissions of carbon compounds (methane = CH_4 and carbon dioxide = CO_2) and secondarily of nitrogen compounds (especially N_2O, not here in a case) that contribute to global warming. Hence, after these deliberations, carbon becomes the last substance case, related to only one problem. The flows of carbon compounds are analysed, with differentiation between emissions of CO_2 and CO_2-forming substances on the one hand, and methane emissions and methane-forming emissions on the other. Here too, the flexible response strategy combines the instruments applicable.

Part Five ends with conclusions on the cases.

5.2 THE USE OF LCA IN MARKETING: MILK PACKAGING

5.2.1 Introduction

In 1990 CML conducted a study on the environmental effects of different packaging systems for fresh milk (Mekel et al. 1990). The study was commissioned by an industrial producer of polycarbonate in the Netherlands, General Electric Plastics (GEP). The interest of this company was primarily commercial. If a polycarbonate milk bottle proved environmentally attractive it could become a viable option on the market. This market is characterised by a dominant position for one-way cartons with a polythene (PE) coating, a diminishing market share for glass bottles and a few minor types of packaging.

At the time, no standard method for LCA, not even a reference method, had been developed. The modest cost of the study, including CML's charges and GEP's internal personnel expenses were still quite high in terms of the limited likelihood of securing the small market for refillable polycarbonate milk bottles. The high level of recycling to be expected and the low weight of the polycarbonate bottle limit the material turnover. However, GEP had several secondary interests. One higher order interest was to investigate how such an environmental assessment would work in practice. Another interest was the signal function for other markets, both milk bottle markets outside the Netherlands and the bottle market for drinks other than milk. The latter market, a huge one, is now dominated by bottles of PET, PVC and glass, cans of aluminium and steel, and cartons, also there.

This case study is *not* one about executing LCAs, however interesting that subject may be. The aim of the case study is to show how (the not yet existing) standard methodology for LCA might function in private activities in society, here in the marketing of materials. The LCA case is described only in brief, since it is not the prime interest here. That is the description of the marketing process. The case study formed the basis of course for these GEP marketing activities.

First, in section 5.2.2, the results of the analysis are given, with a comparison of polycarbonate bottles with glass bottles and carton packs. This section is based mainly on Mekel et al. (1990). The quite short study executed for GEP has been revised here, to bring the classification part more up to date, (but not fully so). Further mainly small changes relate to product specifications, the data used, and the items in the environmental profile. The sensitivity analysis has also been extended. Results remain very similar to the original study. A warning should be given here on how to interpret the results. These results were the best available at <u>that</u> time. Some of the revised process data used originate from a study already completed at the end of 1991. With the current lack of methodological rigour and data on basic processes, any LCA is a preliminary one, but the GEP study , being limited and old, is certainly particularly so. Since at least two studies on milk packaging are now under way (one in Sweden and one in the Netherlands) both of which will improve on our study, it would be a mistake any longer to base practical conclusions in comparing alternatives in milk packaging on the results given here. See further Guinée et al. 1993b on a broader comparison of milk packaging studies that have been executed.

Next, the further societal functioning of the results of the original GEP study is described in section 5.2.3. It is the purport of this case study. The political interventions into this marketing process have broadened the analysis somewhat.

5.2.2 Results of the LCA on milk packaging

The GEP study is described only in brief. The study has not been executed according to the rules specified now in the "Manual for the environmental life cycle analysis of products" (Heijungs et al. 1992b), further referred to as the "Manual for LCA". Its results nevertheless indicate how the life cycle analysis may be used in practice. The data have been updated with the addition of information generated in a case study on window frames (Hoefnagels et al. 1992). The software used is a slightly revised in-house version of that developed for the same study[1]. The results are presented by means of the general framework for LCA given in Part Four.

Goal definition

The LCA study (as against the broader marketing goal and other goals of the commissioner) had a dual goal, first to optimize a polycarbonate bottle for fresh milk and secondly to compare this optimized bottle with the two main alternatives, glass and coated carton. The comparison was for the Dutch situation as envisaged for the near future and not for Europe as a whole. The functional unit chosen was *"packaging of 1000 l of fresh milk, in one litre containers, for household use, in the Netherlands, in the Nineties"*. A better but laborious choice would be "drinking 1000 l of fresh milk in households", the difference being the full life cycle analysis of the milk. Different packaging systems will differ as to amount of milk spilled. Factors determining the differences are the adherence of product residues to the packaging surface, breakage, leaking, etc. For fresh milk the difference between packaging systems will be very limited in the adherence factor. For yoghurt, for example, the adherence factor might be more important. The functional unit chosen abstracts from all such further differences. Only three types of packaging have been included. There are two main omissions in the alternatives investigated. The first omission is the PE bag, with a large market share in Canada, and on the market now in Switzerland, Spain and Germany. The second is the polystyrene cup with aluminium lid.

The containers specified are the result of the design process of aim one. First, the not-yet-existing one litre polycarbonate bottle has been specified, with a weight of 70 grammes. To make an honest comparison, the two other alternatives have been improved as well. The gable top carton presently sold in the Netherlands weighs 31 grammes. A reduction to 28.5 grammes has been assumed. The Dutch glass bottle was 600 grammes at that time. We assumed an improvement to 480 grammes. In the meantime an improved bottle of around 500 grammes has appeared on the market.

Both the lid and the sleeve were specified in a separate design process, the same for the glass bottle and the polycarbonate bottle. Comparing design alternatives is essentially the same as comparing different types of product, requiring a full life cycle analysis. However, in a comparison the elements that are equal may be omitted, making the analysis for design often much simpler. Two elements in the design could be treated

[1] In-house CML software. All computations are by P. Mulder at CML.

separately, the lid and the sleeve. They do not quantitatively influence other aspects of the functioning of the package system. An example of the design process is the lid for both the polycarbonate and the glass bottle. The full life cycle analysis of it revealed that a choice between the aluminium cap and the polythene (PE) cap could not be made on environmental grounds. Both scored better on environmental aspects then operational than the steel/PVC twist-off lid (see Mekel and Huppes 1990, p.17). We chose a re-closable PE lid of 4 grammes. This re-closable PE cap has advantages for the consumer over the aluminium caps now used on glass bottles. With that lid, the polycarbonate bottle also can be transported in the same roll-in containers currently used for the transport of gable top cartons if the bottles are positioned on top of each other. Another analysis showed that the cap, which is four grams, should be included in the deposit system. If not, the material production for thrown away PE caps would completely dominate the results of the analysis of the polycarbonate bottle as a packaging system! The same PE cap, also covered by the deposit, was assumed for the glass bottle. In the mean time, a re-closable glass milk bottle has been introduced in Germany, with a three-gramme PE lid. The sleeve on both bottles is to be a PE sleeve. The sleeves are removed during each trip at the dairy factory. The ink used for the package printing should to removable to allow a high quality level of reuse. See table 5.2.1 for the milk packaging as specified.

TABLE 5.2.1 STANDARD ALTERNATIVES, COMPOSITION[1]

Type	Mass in grammes	
Glass bottle	glass	480
	PE lid	4
	sleeve	2
Polycarbonate bottle	polycarbonate	70
	PE lid	4
	sleeve	2
Gable top	board/carton	25.3
	PE coating	3.2

A high deposit of Dfl.1.- ($0.55) is assumed on all bottles to prevent backsliding in returning the bottles. The choice of reuse frequency, or trip rate could not be based on actual trip rates. Current rates for glass are under twenty five, with large regional differences, according to the private report of one milk filler. The current deposit is only Dfl.0.25, however. The subject of trip rates is much debated in the Netherlands, without the relevant data being substantiated by independent research. With the higher product deposit, the trip rate will rise. We assumed a trip rate of 30 for glass. Experience with the polycarbonate bottle is very limited. The primary comparable use now is as a feeding bottle for baby milk. Technically, the trip rate with normal use may be quite unlimited, and this has been borne out by experience. Like glass, polycarbonate can be heated to well over 100°C for cleaning. We assumed a modest trip rate of fifty, based on a certain amount of inappropriate handling and non-return by the consumer.

[1] See for data on transit packaging Mekel et al. 1990.

Inventory
The full packaging life cycle was investigated, from primary resource extraction to final waste and recycling, including transport, transport packaging such as crates and roll-in containers, bottle cleaning, etc.

The data used in the inventory indicate the possibilities at the time. A full list of all processes is given in Mekel et al. (1990). The process data specified by Hoefnagels et al. (1992) have been added. For LCA software development CML participated in that study on window frames. One important change in the specification of the process tree is that energy is now treated differently. In the GEP study we used the Swedish energy model for electricity consumption in carton production in Sweden. This electricity comes from hydroelectric and nuclear sources, with no environmental interferences specified for these two modes of electricity production. Current computations are based on the European energy model, with a much larger share for coal, gas, and oil. The reason is that a homogenous product such as electricity is traded increasingly over large distances in Europe.

One defect in both the GEP study and the current version is that no separate allocation step has been made. All process data are single process data. Undoubtedly, many of these are "constructed". No explicit method of allocation is given, nor are the basic data available in an non-allocated form. In the current study, the problems of global warming and ozone depletion have been added in the classification. However, all data on ozone depletion are lacking. Data on global warming is very sparse. All process data have been specified in a manner that makes control difficult though not impossible[1].

Classification
There are now more classification factors then in the original GEP study. They are

◇ the use of depletable energy resources ("energy")
◇ units of polluted water (upa)
◇ acid equivalents (ua)
◇ global warming potential (gwp)
◇ ozone depletion potential (odp)
◇ units of polluted water (upw)
◇ solids, partially by mass (solids in kg) and partially by volume (solids in m^3)

These classification categories are <u>not</u> in line with the recommendations in the Manual for LCA. There are important differences. All depletion factors are lacking. Also, human health and ecotoxicity are lumped together, in a medium dependent approach (as upa and upw). Furthermore, several categories are lacking, such as photochemical oxidant formation and over-nutrification. The results of the computations are given in table 5.2.2. The results have not been normalized into the problem contribution of a functional unit such as a "fraction of total problem caused in year" (see Part Four). The results here are very similar to the GEP study, in the categories that remained the same.

[1] All process data have been specified, both in Mekel et al. 1990 and in Hoefnagels et al. 1992, with references to the primary and secondary sources used.

TABLE 5.2.2 ENVIRONMENTAL PROFILE OF DIFFERENT PACKAGING SYSTEMS FOR FRESH MILK. DATA WILL BE SUPERSEDED BY NEWER STUDIES IN PROGRESS AT THE TIME OF PUBLICATION OF THIS STUDY.

Packaging system→ ↓Profile item	**Gable top carton,** 28.5 gr, PE coated, 1 trip	**Glass bottle,** 480 gr, 30 trips, PE lid 4 gr, PE sleeve	**Polycarbonate bottle,** 70 gr, 50 trips, PE lid 4 gr, PE sleeve
energy	742	794	657
upa	183.E3	156.E3	119.E3
ua	459	349	252
gwp	1170	1227	1417
odp	n.a.	n.a.	n.a.
uwp	13108	457	519
solids kg	33	16	19
solids dm^3	0.0	0.0	0.0

Evaluation of scores

There is no formalized evaluation available. The classification results are a good indication of the problems encountered in a non-formalized evaluation. There is neither a dominant alternative nor a dominated alternative. If results are to be arrived at, some *ad hoc* procedure should be applied. By nature, such procedures are debatable. Here are some examples.

If the data on global warming were disregarded, because of incompleteness that leads to arbitrary results now, polycarbonate is dominant over carton. Of course, leaving out classification aspects makes the analysis ever more partial. How does glass compare? In some aspects it is better than polycarbonate (global warming, water pollution, and solid waste). In one aspect it is even worse than carton (energy). If energy takes high priority, glass would become the worst alternative. In Appendix 2 below it is indicated however, that energy depletion is not an environmental problem at all. Then glass cannot be the worst alternative. Without the energy score, or with a very low priority attached to it, the gable top carton alternative would be fully dominated by glass and polycarbonate. If energy receives a priority of about zero (as proposed) and *if* data on global warming are again disregarded, glass is better than polycarbonate in two aspects (water pollution and solid waste) and polycarbonate is better in two others (air pollution and acidification). The quantitative differences are limited. Glass would be superior to polycarbonate if the priorities for water pollution and solid waste are high. Polycarbonate would be superior if the priorities on air pollution and acidification are high. At present, no choice between these two bottle alternatives can be made.

Evaluation: sensitivity analysis

A sensitivity analysis provides the only indication on the robustness of outcomes now available. Technically it involves a change in assumptions, the computation of a new set of data, and a new evaluation. If that evaluation is not altered, it is resistant to the assumptions. Several such hypothetical variants have been computed. Here a main consideration is whether improvements in the product system that might become possible in the near future would alter the results.

For *carton*, the highest improvements result if household waste is burned 100%. One reason for this result is that in the process specifications a high marginal efficiency in electricity production at incineration is assumed, of thirty percent. At incineration, the energy from the wood in carton is transformed into electricity, replacing electricity, mainly from polluting fossil fuels. With the full recycling of carton into carton, also considered an improvement, recycle electricity is no longer produced. Thus, as compared to the standard alternative, net fossil energy use then increases as does the related set of emissions to air. The method used for allocating recycling processes is the cascaded-quality-levels method of recycling, see the GEP study for a description. It is not the value-based method specified here in Part Four. It is not possible now to assess the effects of a change towards this preferred method.
For *glass*, the improvements investigated are the following: the halving of washing energy; no PE label (traditional bottles do not have a label either); the recycling of glass in household waste into high quality clear glass, instead of mixed.
For *polycarbonate*, the improvements in the product system are similar: the halving of washing energy; no PE label; household waste burned 100%, also with thirty percent marginal efficiency electricity production.

How would these changes affect results? Not much, see table 5.2.3. Interestingly, all the orders in scores between the alternatives remain the same. Only glass and carton changed position in terms of the fossil energy resources used, but the difference is negligible (101 vs. 100). Thus both glass and polycarbonate are dominant over carton. The choice between glass and polycarbonate remains as difficult as it was.
It may be noted none of the major changes assumed influence the very incomplete data on global warming contributions, thus substantiating their omission from the current evaluation.

Other assumptions that have been tested are those on trip rate. The differences resulting from changing trip rates are relatively small. A lower trip rate leads to higher recycle volumes and so mitigates the effects of increased material production. However, it appears that all recycling processes are only specified in their positive aspects, through replaced prime material and energy production. With other methods of allocation and more realistic process data on recycling available, the trip rate could become a more important factor.

With all limitations as given, and with many more specified in the GEP study, the conclusion remains that, pending completer research, the polycarbonate bottle is environmentally preferable to the gable top carton. A choice between the glass bottle and the polycarbonate bottle can only be made if specific priorities are set for the different environmental problems. The application of the analysis was the marketing of polycarbonate for milk bottles. That indirect functioning in society is described in section 5.2.3.

TABLE 5.2.3 ENVIRONMENTAL PROFILE OF IMPROVED PRODUCT SYSTEMS OF PACKAGING SYSTEMS FOR FRESH MILK. DATA WILL BE SUPERSEDED BY NEWER STUDIES.

Packaging system→ ↓Profile item	**Gable top carton,** 100% energy recycling	**Glass bottle,** no sleeve, 50% washing energy, high quality recycling	**Polycarbonate bottle,** no sleeve, 100% energy recycling from household waste
energy	547	599	506
upa	101.E3	100.E3	79.E3
ua	236	209	157
gwp	1170	1227	1417
odp	n.a.	n.a.	n.a.
uwp	13108	312	376
solids kg	45	12	15
solids m^3	0.0	0.0	0.0

5.2.3 LCA in marketing[1]

The first step in the further societal working mechanism of the (not yet) publicly available standard LCA methodology was the commissioning of the study by GEP, in the spring of 1990. Since no standard method or reference method was available, nor were supporting tools such as sophisticated software and data bases, this was a daring step by GEP. By using a university institute, the risk that results would not be seen as unbiased was reduced. An independent check on the data supplied by GEP itself, by the B&G Agency, Rotterdam, further improved the neutral status of the results. The study was published in September 1990.

How could the GEP study be used in marketing? At the end of the Eighties the public awareness of packaging waste as an environmental problem was rising. Milk in refillable glass bottles sold mainly at dairies had been replaced by milk in gable top cartons sold at supermarkets. GEP worked out a detailed proposal to replace cartons or bottles with square polycarbonate bottles. The bottles, when worn and taken out of circulation at the dairy factories, would be recycled into high quality polycarbonate for non-food applications. The environmental advantages were substantiated provisionally by the GEP study made by CML at that time. Proposals on the polycarbonate bottle were discussed with several milk factories and retailers.

Why would these parties be interested? The environmental movement was pressing the large retailers towards the (re-)introduction of glass bottles for milk in supermarkets. One guilder deposit PET bottles for soft drinks had already been introduced quite successfully. The public support for such a deposit system was high. That would be reason enough if the system would not be too costly. The aim and reasonable expectation of GEP was that at least one retailer or group of retailers would introduce the polycarbonate bottle, supplied by at least one dairy factory.

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[1] The following description of events is based on extensive talks with all the public and private parties concerned, not just those involved at General Electric Plastics.

However, politics intervened. The Dutch government started discussions with the producers of packaging materials, packagers, and retailers on how to curb the rising flood of packaging waste. The firms concerned united in the Foundation "Packaging and the Environment" or SVM[1], to structure their talks with the Dutch government. The German government was also preparing much more rigid policies on the curbing of waste amounts based on the principle that any producer remains responsible for the waste of his products sold, packaging included. These German developments initiated an interest in the packaging industries to prevent the institution of the same sort of strict measures that threatened in Germany and have been introduced there since. There was thus the pressure of private public opinion pressure "to do something about waste". Secondly, there was Dutch political pressure to do something and thirdly, there was the "*shadow of the German law*".

The largest firms active in the milk field were the dairy factories and some large super market retailers. Both markets are extremely oligopolistic. The two main factories at the time, Melkunie and Campina, since merged, have a comfortable market share of well over fifty percent. The leading Dutch supermarket chain, Albert Heijn, has a market share of over fifty percent for milk retailing. It also has a minority share in Campina/Melkunie. Imports are negligible and form no threat. Shares in the main foreign producers active at the Dutch market have been acquired collectively by the Dutch dairy industry. Apart from the co-operatives, there is one independent dairy factory, Menken van Grieken, with a share of around ten percent. It is owned by the largest soft drink bottler in Europe. At the time, Menken van Grieken was preparing large-scale renovations in its main dairy factory in Wassenaar. The design of the new installation, to cover a limited space, would depend on the choice of packaging system for fresh milk. Menken van Grieken was willing to settle for any packaging that was environmentally worthwhile and cost a reasonable price. It did not want to risk a wrong decision. Thus Menken van Grieken wanted a new and more extensive study on the packaging alternatives, backed by the Dutch Ministry of the Environment. It wanted this backing authoritatively to establish that their choice was the right one environmentally. Without it, there was the chance that the Ministry, in the talks with the dairy industry and the retailers, would arrive at a different alternative. This situation accounted for the first delay in the marketing operations of GEP in the field.

The first reactions at the Ministry were favourable to the study proposed by Menken van Grieken. The Ministry was willing to take part in the study and make a financial commitment if the group of potential commissioners was broadened to include most of the dairy industry. When discussions on the proposal for the broader based study were well under way (with CML as a partner), political developments intervened for a second time. The talks with the Foundation for Packaging and Environment (SVM) were formalized, with the stated aim to arrive at a Covenant on Packaging Waste. In that situation, the Ministry could hardly subsidize a separate study outside the framework being created. It withdrew its support, resulting in the second period of delay. The preparations for the Covenant lasted until May 1991. The Covenant then agreed (Verpakkingen Convenant of 16-5-1991) prescribed certain action but nothing yet for milk packaging. Only a general

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[1] Stichting Verpakking en Milieu.

waste reduction aim was specified, with the option to diverge from that aim if a life cycle analysis supported such a course. This created the possibility for carton producers to prove that their environmental scores were best, or at least no worse than those of the alternatives. The SVM then decided to include milk packaging in the first LCAs commissioned in the framework of the Covenant[1].

If an authoritative methodology for LCA and some publicly supported tools had been available, this delay of several years could have been prevented. That delay is quite likely to increase further as partners with diverging commercial interests fail to agree on the evaluation of results, especially since a formalised ranking procedure with a standard set of priorities is lacking and the relation with the economic analysis is unclear. It is not yet certain even if the polycarbonate alternative will be included in the LCA, since the vested interests do not see this alternative as "viable".

The important thing to note here is that the tool of life cycle analysis was transformed from a private tool of individual firms for the marketing of their products, into a collective instrument with which firms of divergent interests can jointly determine which alternative is best environmentally. GEP still tried to sell the polycarbonate bottle by improving the terms for its clients, e.g. guaranteeing the buy-back of discarded bottles for recycling, and even by financing a large-scale experiment with the polycarbonate bottle. But the Covenant related developments in the Foundation for Packaging and the Environment prevented concrete reactions to any such developments. Especially in an oligopolistic market subject to external threats, the pressures towards conforming to a collective line may be quite strong. The GEP's activities for marketing this bottle in the Netherlands have virtually ceased, except for participation in the current discussions with the Foundation for Packaging and the Environment on how to set up the collective LCA study on milk packaging. The preliminary discussions on defining the organizational set-up of LCA studies have lasted for over a year, from May 1991 till the end of spring 1993. It is beyond the power of GEP to influence Dutch developments any longer. In the political and collective process GEP's private investment in the LCA study it commissioned was lost.

The next important point, however, is that all parties, including the government, have pledged their allegiance to the LCA approach in assessing the environmental effects of packaging. After the delays caused by this pure example of neo-corporatist, horizontal government, the future for the use of LCA in marketing may still look bright. One may expect that collective LCAs, backed by this general commitment will have a firm impact on decision making. This of course will only be the case if clear results in case studies can emerge from the collective process. The declared loyalty to LCA will also enhance the effects of later private studies. Such public support cannot fill the void created by the lack of an authorized standard methodology for LCA with which these private studies have to be executed.

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[1] The study was been requested of TNO, Delft in May 1993.

5.2.4 Conclusions

The GEP example indicates that LCA, freed from the limits set by horizontal politics, will be used offensively, both in product development and in the marketing of products.

The commissioning of an LCA case study is a first step in the societal functioning of a standard methodology for life cycle analysis. That step has been made by GEP, at a time when standard methodology and tools were still lacking.

This state of affairs, combined with the interest-based "horizontal" policy development on packaging in the Netherlands, has so far led to the failure to market the environmentally most attractive option.

5.3 REGULATING SUBSTANCES: THE CASE OF CADMIUM[1]

5.3.1 Introduction

Cadmium is a high-priority target for anti-pollution measures throughout the Western world because of its widespread use and subsequent emissions, its high accumulation in soil, sediment and food-chains, and its high toxicity. In some countries, such as the Netherlands, the pollution problem is so pervasive that the difference between "mean daily intake" and "Acceptable Daily Intake" is very small. Consumer organizations found levels of cadmium in bread from European cereals with a cadmium content nearing the current norm for food. That norm is already deemed too lax. Based on current norms it can be calculated that, if the present rate of accumulation in soils in certain parts of The Netherlands (particularly in the province of South-Holland) continues, many regions may no longer be used for agricultural purposes within a few decades. In all areas where intensive agriculture is combined with a high population density such developments may be expected, outside the Netherlands as well.

There are considerable differences between countries in the way cadmium is regulated. Since the Seventies many countries have introduced quality standards for water and soil, sewage sludge and compost. In some countries these standards were supplemented in the Eighties, with norms and prohibitions relating to industrial products, and with stricter guidelines for production processes. This was the case, for example in Sweden, Denmark, Switzerland, the Netherlands and Germany, but much less so in the remaining parts of Europe and in the United States and Japan. Stricter EC policies on cadmium are on the way. On the basis of an Action Programme on Environmental Pollution by Cadmium and a Council Resolution on this matter (by Ministers of the Environment) on 25 January 1988 major studies have been contracted, and directives are in preparation (Council of the European Communities 1990 and 1990b). Current end-of-pipe policies have been effective in reducing cadmium concentrations in rivers, especially by the much increased treatment of sewage streams. However, total emissions and accumulations probably have not been reduced but only shifted.

In none of the countries, including Sweden and the Netherlands, has sufficient attention been paid to the fact that the supply of cadmium is inelastic. Cadmium is a substance that enters the economy as a trace-element in the ores of other minerals, especially zinc mined for intentional production, and as a contaminant in phosphate ore and coal. Currently, cadmium is produced intentionally only in combined processing with zinc ore. Given current technologies, the amount of cadmium produced depends solely on the amount of zinc produced. If demand for cadmium changes through a prohibition for example, the price of cadmium will change but the amount produced will hardly alter. Nor will all the other flows into the economy because they are unintentional. From zinc ore, some cadmium accompanies zinc as an unwanted contaminant and some goes into the waste streams of zinc production, especially jarosite. These unintentional forms are complementary to the intentional production of metallic cadmium, with the total amount of cadmium being determined by the zinc production. From phosphate ore, the cadmium

[1] This chapter is based to a large extent on Udo de Haes et al. 1990; Huppes et al. 1992; with revised figures based on van der Voet et al. 1993, in press.

either goes into the phosphate produced as a contaminant, and hence into phosphate-derived products and soil, or it goes into the waste streams from ore processing.

Being a trace-contaminant in bulk ores causes additional control problems with cadmium compared to the control of the main elements in the ores such as zinc itself. Large quantities have accumulated in the economy and are to be found for instance in the slag heaps of ore-smelting plants, formed in previous years and even in previous centuries. Thus, measures aimed at reducing the environmental pollution caused by specific cadmium flows, no matter how effective they may be in themselves, threaten in the long run simply to shift or pass on the problems. This applies in particular to recycling measures and product regulation, in short to measures that are currently the focus of attention.

The aim here is first to describe the relevant cadmium flows (5.3.2), to develop a coherent technical approach for tackling the cadmium problem based on specific available solutions (5.3.3), and to develop the flexible response strategy for cadmium, first by surveying the applicability of several policy instruments (5.3.4) and then specifically by working out the substance deposit in more detail (5.3.5). Summary and conclusions on the cadmium problem end this first case study on substance policy (5.3.6).

5.3.2 Substance flow analysis of cadmium in the European Community

In the analysis of flows of such a substance as cadmium a first screening should indicate whether the problems it is related to - depletion, and toxic effects on man and nature - differ strongly between compounds. If some compounds are not related to any problem or, conversely, constitute the full problem, they may either be omitted from the analysis altogether or be treated separately, in the substance flow analysis of that specific compound. Although in some compounds, like enamels, cadmium is extremely immobilized, the analysis given here takes into account all occurrences of cadmium. Metallic cadmium, in itself quite stable and harmless, is transformed in several types of long-term and short-term processes into usually harmful compounds. Therefore no flows can be excluded from the analysis.

The boundary-crossing flows of cadmium in the economy and the environment of the EC are given in the main substance flow diagram, see figure 5.3.1. Table 5.3.1 gives the data on a somewhat less aggregated basis. The total structural inflow of cadmium into the economy in 1987 (and approximately the current situation) was about 11,462 tonnes, with 86% imported and 14% extracted from the lithosphere within the EC. The most important flow is that of zinc ore at 44% of the total inflow into the economy, with 34% through imports and another 10% through extraction within the Community. Second largest is the import of refined cadmium at 20% of the total structural inflow in the economy.
The structural outflow of cadmium from the economy consists mainly of export, and was a bit over two thousand tonnes. The main export flows are cadmium pigments, cadmium in plastics, refined cadmium, and cadmium in batteries. Since structural outflow is thus very limited a high nett structural inflow results, of 9417 tonnes.

FIGURE 5.3.1 GENERAL SUBSTANCE FLOW SCHEME OF FLOWS AND ACCUMULATIONS OF CADMIUM IN THE ECONOMY OF THE EC, 1987, IN TONNES

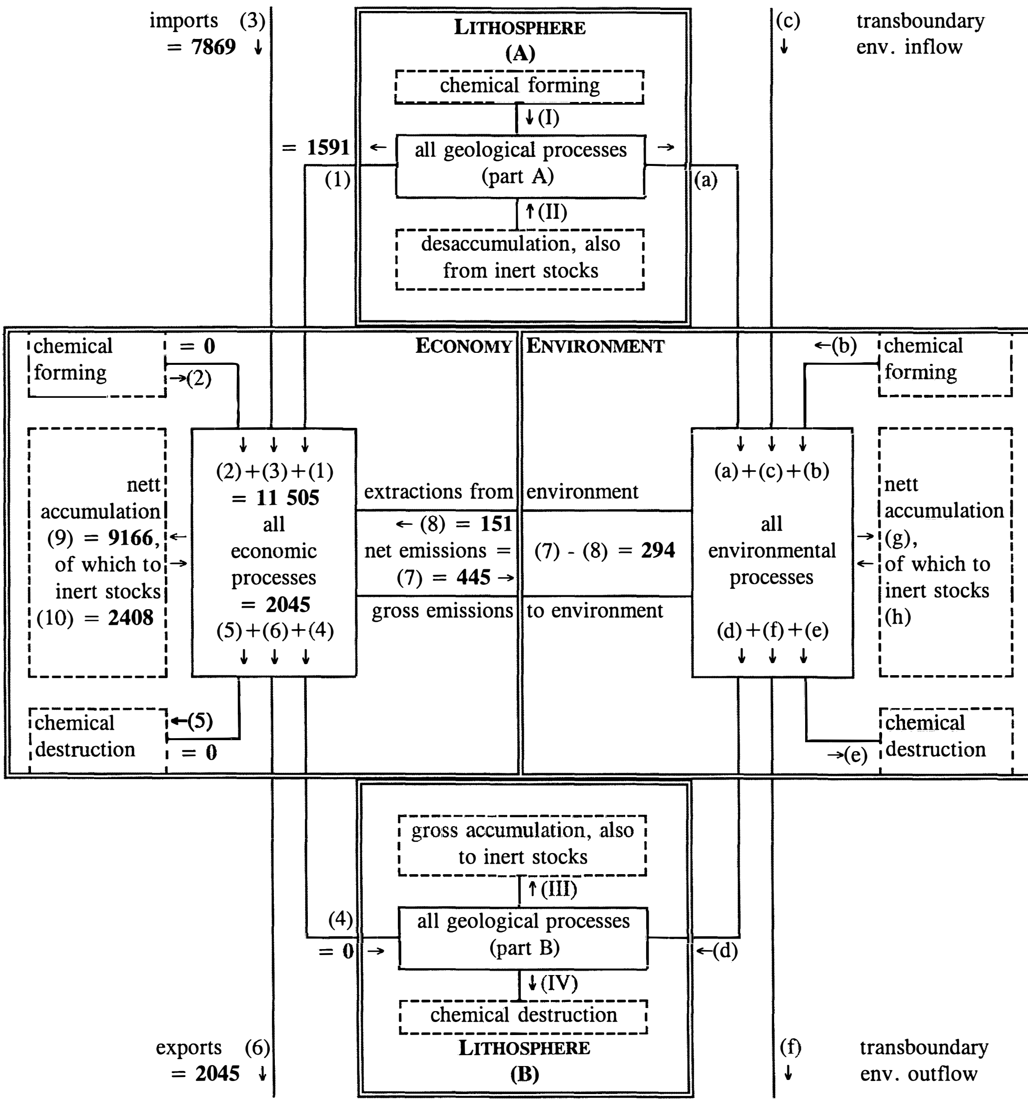

TABLE 5.3.1 FLOWS AND ACCUMULATIONS OF CADMIUM IN THE ECONOMY OF THE EUROPEAN COMMUNITY, IN TONNES PER YEAR, 1987

1[1]. Extraction from lithosphere		4. Deliverance to lithosphere	
fossil fuels	55	None	
iron ore	2		
zinc ore	1189		
marl for cement industry	345		
Subtotal from lithosphere	*1591*	*Subtotal to lithosphere*	*0*
Subtotal to lithosphere	0		
Net inflow from lithosphere	**1591**		
2. Chemical forming in the economy		5. Chemical destruction in the economy	
None		None	
Subtotal chemical forming	*0*	*Subtotal chemical destruction*	*0*
Subtotal chemical destruction	0		
Net chemical forming	**0**		
3. Imports		6. Exports	
refined cadmium	2321	refined cadmium	323
batteries	2160	batteries	313
stabiliser	212	stabiliser	420
pigment	16	pigment	468
miscellaneous products	9	miscellaneous products	132
fodder	2	fossil fuels	18
fossil fuels	76	phosphate fertilizer	34
iron ore	18	phosphoric acid and similar	85
non-phosphate fertilizer	43	zinc ore	173
phosphate fertilizer	256	zinc	63
phosphoric acid and similar	34	zinc scrap	16
phosphate ore	511		
zinc ore	3867		
zinc	340		
zinc scrap	6		
Subtotal imports	*9871*	*Subtotal*	*2045*
Subtotal exports	2045		
Net imports	**7826**		
Total structural inflow[2]	*11462*	*Total structural outflow*	*2045*
Total structural outflow	2045		
Net structural inflow	**9417**		

[1] Numbers correspond to those in the flow scheme.

[2] *Total structural inflow* is (1) + (2) + (3). *Total structural outflow* is (4) + (5) + (6). *Net structural inflow* is total structural inflow minus total structural outflow .

8. Inflow from the environment		7. Outflow to the environment		
Dredged sediment	66	To ground water		93
Deposition on agricultural soil	85	agricultural soil	93	
		To surface water, inland		141
		industrial waste water[a]	58	
		landfill	24	
		sewage water treatment	54	
		agricultural soil	5	
		To surface water, seas		50
		sewage sludge	3	
		waste treatment sludge	22	
		P-refinery waste gypsum		25
		To air		152
		cadmium alloy industry	1	
		battery industry	11	
		sewage water treatment	2	
		household waste treatment	20	
		cement industry	7	
		agricultural soil	35	
		energy production	35	
		iron refinery	18	
		zinc refinery	23	
		To non-agricultural soil		9
		products	9	
Subtotal from environment	*151*	*Subtotal to environment*		*445*
		Subtotal from environment		151
		Net emissions		**294**
Total inflow into the economy:		Total outflow out of the economy:		
*from lithosphere	1591	*to lithosphere		0
*chemical forming	0	*chemical destruction		0
*imports	9871	*exports		2045
*from environment	151	*to environment		445
Total in	*11613*	*Total out*		*2490*
Total out	2490			
Accumulation on balance	**9123**			
9. Accumulation in the economy *from balance:*	*9123*			
Mobile stocks				
in products and installations[a]	6306			
in agricultural soil[a]	406			
Inert stocks (10)				
in landfill[a]	2403			
in strategic storage	5			
Subtotal	*9120*			
Rounding errors	3			
Total accumulation	**9123**			

[a] is balancing item

The nett structural inflow partly accumulates in the economy, at 97%, and partly results in net emissions to the environment at 3%, leading to what are already dangerous levels of cadmium. The discomforting conclusion must therefore be drawn that over half the current net structural inflow accumulates in mobile stocks in the economy. This part of the cadmium accumulation consists mainly of cadmium in batteries and in plastics and in waste stored in the sites of industrial premises, as "waste to be processed". Since the lifetime of most cadmium containing batteries is only about five years, the emission of cadmium from these batteries will increase dramatically in the near future, unless proper countermeasures are taken. The other part of the accumulation is in landfill, here declared to be "immobile stocks". Many of the wastes in landfill are better seen as emissions to soil.

The nett structural inflow of cadmium in the EC, at 9417 tonnes, is 29 grammes per inhabitant per year. This compares roughly with the amount of net structural inflow in the Netherlands (354 tonnes, or 25 g/inhabitant), based upon independent and more elaborated statistics (van der Voet et al. 1989).

5.3.3 Technical solutions

As already observed, measures for the emission reduction of cadmium can be focused on a decrease of the inflow, an increase of the outflow, and on emission reduction itself. The three are not independent: inflow minus outflow is nett emissions. In long-term equilibrium the accumulation, at least in mobile stocks, will be zero. Fundamental policies will result in effects at the system boundaries, with reduced emissions resulting from a decreased net structural inflow. Such policies work at the boundaries of the economy system. Processes and products may also be influenced within the economy, with these boundaries affected only indirectly. Since the supply of cadmium is inelastic, such measures on the level of individual processes and products within the economy, if reducing emissions at all, will primarily lead to increased outflow. Given the inelastic supply and the large accumulation in mobile stocks that has taken place in the last decade or more, measures directed at increased outflow to substrate seem indispensable. For individual countries and the EC as a whole, it is possible to realize internal improvements by exporting the problem. This cannot be a permanent solution. Either it will evoke countermeasures or it will lead to the ethically unacceptable consequence of pollution in administratively less able Third World countries (now that the Second World will no longer be available). This does not mean that all export of potential emissions should be halted. That is a separate subject not to be treated here. It means that measures are acceptable from a substance policy point of view only if the improvements are *not* realized by a deterioration in other countries.

Inflow related measures

The inflow of cadmium into the world economy is not so much regulated by the demand for cadmium itself, as by the demand for the ores in which it is a contaminant only (phosphate ore, iron ore, coal) or in which it is present in concentrations with a much lower value than other elements (zinc ore). Cadmium for intentional use now is produced only in combination with zinc production. For a given technology, co-producing cadmium is almost costless. It has to be removed from the zinc anyway for technical or current policy reasons. At a global level, decreasing the cadmium inflow thus can be achieved only by decreased mining and refining of ores, especially those of zinc and phosphate.

Reduced use of zinc and phosphate in the EC, or any country, thus leads to a reduced inflow and emission of cadmium in the whole world. By contrast, reducing its import, as a metal or worked into products, hardly has any effect on the global inflow at all. It thus leaves the rest of the world with a larger amount of cadmium to get rid of, in the end. A combination of techniques used in the EC, or any country, might work out neutral for the rest of the world. This might be stated as an aim for cadmium policy. In concrete terms, it means that the reduced import plus the increased export of cadmium resulting from certain measures, are allowable to the amount of cadmium *not* produced by other measures, i.e. by reducing the consumption of zinc and phosphate requiring products. To make this criterion operational further specifications are required as to what, in this context, constitutes a reduction in consumption. How much phosphate use is implied in eating meat from domestic pig farming, partially based on foreign fodder? The operation of this measure will be discussed later.

Reduced inflow of zinc

Zinc is used for plating (50 percent), as a metal (40 percent), and in miscellaneous products including paints, rubber, etc. (10 percent). The replacement of zinc by aluminium and plastics is already taking place, and could be enhanced if these alternatives are made economically more attractive[1]. Zinc is also being reused for plating and as a metal. The total reuse and substitution potential of zinc is estimated to be about 30-40 percent at current prices, and could increase to about 70 percent if it becomes economically more attractive. Possible measures include

◇ replacement of zinc plating by aluminium and other coatings
◇ replacement of zinc as a metal by plastics and other metals
◇ replacement of zinc in miscellaneous products such as rubber and paints
◇ increased reuse of zinc in plating and as a metal.

If zinc production were to stop the only current source of intentionally produced cadmium would disappear, as would most contaminants in products and wastes, and finally, most emissions. The present pollution problem would be replaced by the depletion problem of increased scarcity of cadmium as a raw material. Zinc production might diminish or even stop without any policy measures, because stocks of the ore become depleted. The question then arises: does it make sense to reclaim cadmium after collection for possible future use? The cadmium study by Udo de Haes et al. (1990) directed attention to this question. Of central importance are the estimates of the reserves of zinc ore. It is difficult to get hold of data on the stocks of such an economically important and scarce raw material as zinc. It has been pointed out repeatedly that zinc stocks could be exhausted within the foreseeable future. The Dutch governmental report on the state of the environment 'Concern for Tomorrow' (1991) states that zinc ores will last only the extremely short period of fifteen to twenty years, according to a survey of studies. However, Vonkeman, in 1974 presented a table based on the Report to the Club of Rome (Meadows et al. 1972). Eighteen years ago it specified the period of eighteen years before exhaustion of proved zinc stocks, at the rate of use of that time. During this period zinc

[1] The use of aluminium and plastics may have broader environmental disadvantages. Before any replacement measures are taken these disadvantages should be evaluated, preferably on the basis of LCAs.

ore mining and zinc use increased substantially. Proven stocks, however, remained the same in terms of years of current exploitation.

Several mechanism prevent realistic insight into the future availability of zinc ore. If amounts marketed decrease, its price will increase. This in turn will lead to the extra availability of ores, both because exploration will be intensified and because known poorer reserves will become profitable. Moreover, the amount of proven stocks might well be related more to the number of firms extracting the ore than to any underlying technical and economic reality. The expense of exploration can only be justified by the exploitation of the reserves found within the reasonably near future. With a small number of firms active in zinc ore mining, the expectations on their long-term sales would guide the planning of exploration. A planning horizon of eighteen years is quite long for most firms in general, but probably not for firms in an oligopolistic resource market, where the development of a mine may well take more than a decade. Real insight into reserves thus being lacking, zinc - and hence cadmium - might in fact be one of the resources threatened with early exhaustion. Let us assume an availability for not more than 50 to 100 years, at current price levels. Both for pollution reasons and for reasons of zinc depletion it thus might make sense to produce less zinc. For potential depletion reasons of cadmium, it also might make sense to store cadmium for future use.

Reduced inflow of phosphate

The environmentally problematic cadmium emissions from fertilizer and manure have their origins in the import and processing of phosphate ore and in the import of fodder. Reduced inflow may be realized by higher efficiency in the application of phosphorus and in its substitution for other substances, as with some efficiency enhancing enzymes in fodder now marketed in the Netherlands. Several other methods for increased efficiency are technically available, primarily in animal husbandry, and the processing of manure before its re-application in the production of food and fodder. On the inflow side, an obvious measure for reducing cadmium inflow is to import and process only cadmium-poor phosphate ores, leaving phosphorus import intact. See the case study on nitrogen and phosphorus in the next chapter for more details on this point.

Outflow related measures

The export of cadmium is an economic outflow of cadmium from the EC. If not bound to increased imports it implies reduced cadmium emissions in the EC and higher levels abroad. Encouraging cadmium export, however, is not regarded as an acceptable measure to diminish cadmium problems within the EC. The outflow-related measures should thus aim primarily at the increase of the "acceptable disposal" of cadmium, either back to the lithosphere or through the accumulation of really inert stocks in the economy. Increasing acceptable disposal means that economic flows and waste streams must be controlled, and disposal must not result in emissions to the environment. Criteria for what is meant by "acceptable disposal" have not yet been formulated[1].

[1] From an environmental point of view one could say that leaching times then should be measured on a geological time scale, taking into account cumulative emissions during at least several thousand years.

Some forms of acceptable disposal, also of newly recovered cadmium, might be attractive because of the long-term depletion problem. The zinc supply is not endless. This means that the cadmium production from zinc refining will decrease at a certain time in the future. Decreased zinc use due to reuse and replacement of zinc as described above will expand the time scale during which zinc may be mined and virgin cadmium may be extracted. When zinc ores are finally depleted, however, the stored cadmium could then be used. Disposal now thus becomes a strategic economic stock. It is an open question if there will ever be a serious depletion problem related to cadmium. Demand will decrease because substitutes for the high priced cadmium will be developed that may become economically attractive in themselves. At a higher price, with only smaller applications where cadmium is extremely essential, the extraction of cadmium from other sources, such as phosphate ore and coal fired energy production, becomes economically feasible. Possible measures include

◇ immobilization of cadmium (together with other potentially hazardous substances) in waste streams by chemical and physical means.
◇ disposal in leak proof sites from which cadmium cannot be recovered "back to the lithosphere". An example is in disused mines or in plastic and in concrete in growing silt deposits at the bottom of the sea.
◇ controlled disposal at sites from which cadmium can be recovered. An example would be in well guarded closed sites for chemical waste.

Emission reduction

The reduction of cadmium emissions by means of effluent treatment measures, such as flue gas cleaning, can be an effective way of reducing a particular emission, but will always lead to an alternative emission, to a new waste stream, or to reuse in some product that ultimately will become a waste itself. Application of these measures by themselves is therefore insufficient to effect emission reduction. To be effective, effluent cleaning must always be combined with a process technique that creates an outflow. It should therefore be in a package of measures with one of the three types of outflow listed above. Although some emissions, especially of a local nature, are less risky environmentally in the short term, a systematic differentiation between types of emissions does not seem to make much sense in the long run.

Recycling

As noted above, reducing cadmium emissions should be linked to a decrease of the cadmium inflow, and/or an increase in the outflow of cadmium to immobile stocks. Recycling of cadmium might at first be seen as an emission reduction. It is not an emission reduction, not directly at least. If it is to become a reduction, it should either be combined with an inflow reduction or an outflow increase. Given the nature of cadmium as a contaminant and as a product with an inelastic supply, the recycling of cadmium will not lead to a reduction in inflow. This is the case even if recycling would lead to a substantial fall in prices. Nor are there special reasons why recycling would lead to an extra outflow. On the contrary, recycling will be from concentrated uses of cadmium where separate collection is relatively worthwhile. By recycling, the total supply of usable cadmium is increased. This will lead to an extra use in all types of applications, including diffuse forms, and not especially in those that may be recycled. Thus recycling will lead to the extra application of cadmium that cannot easily be collected separately. Hence, recycling of cadmium will lead to extra emissions of cadmium. In the short term, recycling may lead to an emission reduction, due to the extra accumulation in the economy. This effect is only temporary, until the leaks to the environment from all applications have grown enough to reach a new equilibrium with the constant inflow. In the longer term, the recycling of cadmium will inevitably lead to an increase in the emissions of cadmium.

Recycling techniques therefore are no solution to the emission problem for cadmium[1], nor for any future depletion problem. The uses of cadmium with separate collection should be stimulated, as would be necessary for recycling as well. Rechargeable batteries, and bright coloured PVC, PP or PE crates for beer and soft drinks, with cadmium pigments and stabilizers, are major examples of readily collectible products. After collection, the cadmium should not be recycled but transformed into a technical outflow, not an outflow in the form of an export. As a consequence, non-collectible applications, such as small or built-in plastic parts or products, or cadmium-containing paints will then decrease. However, before "collectible" effectively leads to "collection" an adequate collection system should be developed and operated for such cadmium-containing products. Collection and storage may also be implemented for diffuse waste flows such as household waste incineration and sewer and industrial waste water purification. Current levels of expenditure on such end-of-pipe techniques indicate that they are extremely costly, see the critical tax levels in table 5.3.2[2].

5.3.4 Policy instruments

This section begins with a specification of the instruments applicable to several streams, with as much emphasis on more macro instruments as possible. In the next section the flexible response strategy applying to cadmium is drawn up. There will be a certain quantification of instruments and effects.

Which instruments might be applied to the different flows of cadmium? Stepping down the ladder of freedom and efficiency, the following possibilities are available:

Structural
- * (Extended liability)

Cultural
- * Standard methodology for LCA

Financial
- * Substance deposit
- * Uniform emission tax
- * Estimated emission tax

Prohibiting
- * General emission design standard
- * Estimated product standard, LCA based
- * Estimated emission design standard

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[1] Recycling of materials with a more elastic supply may of course lead to a reduced inflow. For zinc, recycling is very sensible, as indicated.

[2] In the Netherlands the sewer sludge cannot be used in agriculture any longer mainly because of contamination with heavy metals. The sludge thus becomes a waste. Political reasons make it very difficult to find dump sites. All these wastes are going to be burned at an estimated cost of around five hundred million dollars a year. It seems hardly imaginable that these costs alone are justified economically by the polluting applications of heavy metals as the main contaminants. Besides, the cleaning operations contribute to several other environmental problems.

Structural instruments

Extended liability is applicable in principle to local soil pollution. If effective, it leads to shifting the cadmium flows to other products and waste streams, including those with more diffuse emissions to which even extended liability is not applicable. The net effect on potential environmental harm done might well be negative since local pollution is probably preferable to the diffuse pollution that might be its alternative. Only a very partial application of liability is possible. Partial application would thus seem to have the same disadvantages as any measures taken at a micro level. Even if effective at the partial level, the effects at the higher macro level are much more limited and in the case of cadmium may even be negative. The specific location of local soil pollution, however, may also be influenced, reducing the chances of especially high risks and costs. Special liability rules for cadmium do not seem to make sense.

Cultural instruments

The standard method of LCA would include all regular flows in products related to cadmium. Problem-shifting to other substances may thus be prevented. The more thorough analysis of one substance in a substance flow analysis, especially combined with an economic analysis, may better indicate indirect effects than LCA can do for all the substances concerned in a product. The non-use of cadmium will come out of an LCA as a better alternative, while the inelastic inflow may imply that such a shift will increase emissions. Is there place for other types of information? Information and education may influence two different types of choices, such as changes in marketing and purchases and changes in behaviour at disposal. Changing market behaviour according to SFA-based information only on cadmium would not seem an effective option for several reasons. Since products are related to many more environmental problems through extraction and emission of many more substances, general product information is preferable to a dazzling array of individual substance analyses. The substance-product information cannot be combined with this general product information. Furthermore, because cadmium has an exceptionally inelastic supply, a change by some consumers to cadmium-free products will lead to exceptionally high secondary effects on cadmium itself. The price drop in cadmium will go on until all cadmium has found its way into products again. Not buying nickel-cadmium batteries now may in many circumstances lead to higher or more hazardous forms of emissions elsewhere. However, knowledge of the dangers of cadmium may support any separate collection scheme. It may also indicate the dangers of recycling cadmium. Especially in branches where larger amounts of cadmium are used, firms might support schemes that lead to a separate collection, as is currently the case with some larger nickel-cadmium batteries (e.g. of Bosch). Separate collection, however, involves the danger of profitable recycling. The nett effect of cultural instruments is dubious, because of the opposed results of individual behaviour, as e.g cadmium-free batteries and the nett collective effects thereof, extra emissions.

Financial instruments

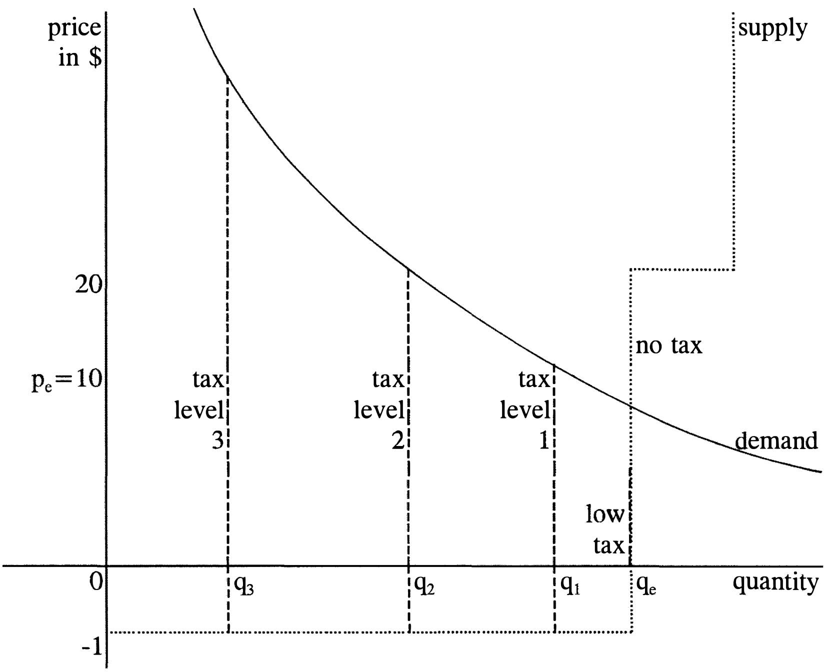
The substance deposit seems the best instrument for curbing cadmium emissions and for limiting depletion through storage. At the global level, there is only a limited number of mining processes that bring cadmium from the lithosphere into the economy. The outflow will only be organized in an environmentally effective manner if a stimulus towards specified techniques of immobilization and storage is given. The details of application are worked out in the flexible response strategy in the next section. The effects on inflow would be limited, even of extremely high deposits. The main effects would be through increased outflow in environmentally acceptable ways.

Emission taxes are applicable only to a limited number of streams, especially to large installations in ore processing, in energy production and in waste incineration, but not in most waste dumping. If applied only where applicable, at a suitable level of tax, the following developments could take place. In ore processing, the production of metallic cadmium will increase to avoid the emission tax. In energy production, energy intensive cleaning technologies will be introduced, eventually leading to metallic cadmium as well. In waste incineration the same thing might happen. Some preventive effect might result if the originators of the cadmium containing waste are known and accountable. Other sources are difficult to monitor. This is true especially for household waste and sewer streams. As a general result of the only very partially applicability of the emission tax alone, substantial problem shifting may be expected.

Some second-choice financial instruments are discussed in the political arena. Product taxes, if proportional to the amount of cadmium in the product, might be an easy proxy for the difficult-to-apply emission tax. If product taxes are applied broadly and equally, a system for taxation of cadmium would in effect result, instead of taxation of cadmium emissions. Given the inelastic supply, these taxes will be borne primarily by the producers of cadmium. There will not be much effect on the amount produced, because of the inelastic supply. Nor will there be any influence on the way these products are treated as a waste. Within a wide range of tax levels the environmental effects will be minimal. The real costs, primarily the costs of implementation, will be limited as well. If product taxes (or the deposit) are set high enough, the product market for cadmium will cease to exist and all cadmium would become waste. Highly profitable applications of cadmium would not then cease to exist, they would be based on waste-recycling. See figure 5.3.2. Current equilibrium price is at p_e, at around \$10. At a negative price level of \$ -1 it becomes more profitable for zinc producers to bear the costs of storing cadmium as a chemical waste, for about \$1 per kg. If demand were to increase, raising prices to \$20 for example, new techniques for cadmium extraction from waste streams would become profitable and would be installed, shifting the supply curve to the right.

Introducing the all-pervasive product tax equivalent to less then \$10 per kilogramme of cadmium, i.e. the current price, would have scarcely any effect at all. The price slides down along the supply curve, without any material changes in the processes concerned. The tax would effectively be paid by the zinc producers. At tax level 1, around \$12, real effects start to occur. High costs to the producers, because of the transfer payments involved, will be associated with very limited effects, the shift from the tax-free equilibrium quantity q_e to q_1. That amount of cadmium will now have to be treated as a waste, at an assumed cost of \$1 per kilogramme. This peculiar situation implies that cadmium producers are willing to supply cadmium not only for free, but even with a bonus, to avoid the costs of waste processing. Real costs will be low and emission reductions debatable, depending on the type of waste processing. With a movement to tax levels 2 and 3, more substantial real shifts start to occur. Cadmium is taxed off the market and worked into waste streams. Regulations on these waste streams, not influenced by product taxes at all, will decide whether the nett effects are environmentally

FIGURE 5.3.2 THE EFFECTS OF CADMIUM TAXES OR CADMIUM-BASED PRODUCT TAXES ON THE MARKET FOR CADMIUM



attractive or not. The high costs of consumption forgone are thus associated with no specific environmental effects.

The equal application of product taxes in terms of emissions caused or prevented, a condition for both justice and efficiency and assumed above, is next to impossible. It seems very difficult to tax any amount of cadmium used at least once. Taxed once, however, it seems unavoidable that some flows will be taxed twice or even more often. Recycled cadmium from, and into taxed products would be taxed twice. Products in a chained production process, such as batteries built into household appliances, would be taxed twice. And if such appliances would be recycled, they could be taxed three times. If emission taxes on incineration are also involved, the same amount of cadmium might be taxed a fourth time. Any rational manner for minimizing the costs of emission reduction would soon become impossible.

Prohibiting instruments

General emission design standards to reduce cadmium emissions at a micro level must be supplemented with product regulations to be effective at all. They would have to go to extremes if, at a macro level, they are to be effective. The situation is similar to the second-choice financial instruments. The changes described there could be implemented by direct regulations. Problem-shifting abroad would necessarily occur, especially to Third World countries that do not have the administrative apparatus to implement comparable countermeasures. This is at variance with the requirement that solutions should not imply problem-shifting to third countries. Direct regulations thus cannot be as effective in preventing emissions globally as can the substance deposit here the main financial instrument. Only if all non-controllable applications are forbidden and if all remaining waste streams are controlled, will the large emission reduction desired at the EC or national level occur, based to a large extent on problem-shifting to abroad.

5.3.5 *The flexible response strategy for cadmium: substance deposit*

The main role in the flexible response strategy is played by the substance deposit. There could also be a minor role for information. The substance deposit on cadmium means a deposit payment on all economic inflows of cadmium, that is on import and on extraction from the lithosphere, and a refund on all economic outflows, that is on export and on "back to the lithosphere".
Import consists mainly of a limited number of bulk flows and concentrated flows, in zinc ore, zinc and refined cadmium, in phosphate ore and phosphate, in other bulk resources (e.g. coal), and in nickel-cadmium batteries. Domestic extraction consists mainly of zinc ore and limestone. Together, these imports and extractions account for over 95% of all inflow. The last 5%, around 600 tonnes, should be approached in a practical manner. Where flows in imports and resources extracted are recognizable the deposit should be paid. All such imported products involving a negative effect on competitiveness should be taxed as well, to prevent undue commercial disadvantages for specific firms within the EC. The famous cadmium-red button on imported typewriters might be left for what it is. It is quite irrelevant both environmentally and economically. For reasons of equity such applications might be brought under the deposit system (cadmium pigments may be readily recognized) or may be forbidden.

Deposits must be refunded upon the export of all cadmium containing products and waste streams, and upon disposal of waste streams back to the lithosphere. Exports consist of the same types of items as imports, in different quantities. What is "back to the lithosphere"? There are extreme cases, such as depositing the wastes in coal mines. For a product such as cadmium storage at a long-term, stabilized landfill site might also be seen as a flow back to lithosphere and thus form the basis for refunding. Certain inert stocks in the economy, if they can be readily kept in that state, might also be treated as an outflow for all practical purposes. The strategic storage by governments in the form of an inert stock for possible future use, might be seen as such an outflow with a right to refund. Storage in a long-term stabilized landfill site could be seen as a relevant inert stock in the economy, not as a flow to lithosphere. Exact specifications are required for such refundable flows to the lithosphere and, possibly, to inert stocks in the economy, determining on which such activities a refund will be paid. A still more flexible but environmentally more risky approach is to rate guarded dumps and certain applications as in enamel, according to the percentage that might leak in the very long term. Refunding would then only apply to the amount not leaked out. Such approaches, however, could lead to another type of leak, a financial one. Cadmium worked into enamel might earn a refund of say 50%. It might then be freed from its chemical bonds by burning and then be worked into enamel again to earn the refund for a second time, etc. The loss in net proceeds through such fraud might be a nuisance; the incentive not to emit would remain fully intact.

Deposit/tax amount

It is expected that in the near future the amount of cadmium in waste will increase substantially due to the recent increases in the use of nickel-cadmium batteries. A deposit level for cadmium should therefore be at least high enough to cover the costs for the collection and acceptable disposal of batteries. The most common type, the penlite, should be recovered almost entirely. Larger batteries would then be fully recovered as well. A higher percentage of the still smaller batteries will be lost. Suppose that separate collection requires real costs of ECU 1000 per tonne of batteries and that the costs of acceptable final disposal are taken extremely high as well, at ECU 2000 per tonne of batteries, including transport. The total real costs of collection, transport and disposal amount to ECU 20 per kilogramme of cadmium, based on 3 grammes of cadmium per penlite of 20 grammes. These costs of disposal are about twice as high as the current market price. Also suppose that most consumers will bring back their nickel-cadmium penlite batteries if they receive a high refund of 0.5 ECU apiece. That is around ECU 165 per kilogramme of cadmium. Refunding should at least cover both these costs of ECU 20 and ECU 165. A high deposit level deposit would thus be ECU 185 per kilogramme. Non-collectible applications would mostly disappear. The level should be that high to induce widespread separate collection for refunded outflow, in this case not for recycling but for safe disposal.

See table 5.3.2 for some other critical tax levels for emission-reducing measures. Current purification costs are extremely high. These costs have been attributed to cadmium alone however. Let us assume that, somehow (see the section on allocation in Part Four, chapter 2), the cadmium share of total costs could be set at 10%. These allocated purification costs are still extremely high, at ECU 2000 per kg cadmium. End-of-pipe cleaning procedures seem an unwise option to pursue in comparison to prevention. Such choices need not be made under the deposit scheme proposed. The cheapest options will, after some time, be worked out in numerous decisions. If prevention pays, it will be prevention, if cleaning is cheaper cleaning will be preferred.

Administrative application

As long as the deposit system is set up to closely resemble the customs and excise systems in the different countries, there will be little difficulty in getting it to operate efficiently, whether applied to imports and exports, or to internal EC production. The substance deposit is to be applied to virtually all cadmium flows:

◇ For controlling the major imports and exports of cadmium specified, the administrative system (based largely upon customs) is virtually already in place.
◇ For controlling cadmium extraction from the lithosphere, inside the EC, mainly zinc ore and limestone, the excise system already existing for alcoholic beverages and car fuels, can easily be extended, depending upon the application.
◇ For refunding certain deposits, criteria will be needed to define exactly what constitutes acceptable disposal. The environmental authorities currently responsible for monitoring existing standards could assume the additional inspection responsibilities, with the necessary additional personnel, although any deposit refunds would have to be dealt with by the relevant excise officials.

TABLE 5.3.2 CRITICAL TAX LEVELS (CTL, IN ECU PER KG CD) FOR EMISSION REDUCING MEASURES[1]

Measure/technique	Critical tax level
Reduction of industrial water emissions, non-p-industry[2]	24000
Reduction in sewage effluent[3]	20000
Sludge in depot[4]	2000
Emission P-industry[5]	416
Zinc scrap recycling[6]	116
Replace zinc parts by plastic[7]	141
Low Cd P-ore	?
Replace nicad batteries	?

The most straightforward manner in which to determine the administrative costs of the proposed system for cadmium is to estimate the number of new administrative transactions which will be required by the system, and to multiply this volume of work by the approximate cost per transaction. Table 5.3.3 presents the maximum number of transactions required at the current volume of cadmium flows through the economy. The main part of operations is required only for a minor flow. "Other materials" and "other products" require two-thirds of all transactions, for about 2% of the inflow and 10% of the outflow. After the behavioral adjustments due to the cadmium deposit, the number of transactions required will drop substantially. Moreover, the number of transactions with an extremely low amount of cadmium involved may be reduced further by either leaving those flows out of the deposit system or through forbidding these applications. The total number of transactions required would not exceed 300.000. If the costs per transaction are set high, at ECU 150, that is three times the current average amount in customs and excise operations, total administrative costs would be about ECU 50 million per year, using the existing customs and excise offices. That European total is a small fraction of the costs, in the Netherlands alone, of the sewer sludge burning being introduced there because of its contamination with heavy metals. If a deposit were introduced for several substances, the additional costs of administration will drop. If a deposit already exists for phosphorus and is applied to the import of its ore, the costs of including the cadmium in this administrative step will be negligible.

[1] Source: Huppes et al. 1992, p.149.

[2] Includes all other metals.

[3] Includes all other metals; supposedly 50% has been removed already.

[4] Costs are 0.63 ecu/kg N+P = 0.11 ecu/kg P = 2000 ecu/kg Cd (Cd content P-ore 50 mg/kg P).

[5] No indirect effects on other emissions and other costs for emission reduction taken into account.

[6] Costs are ECU 0.07/kg scrap recycled into zinc.

[7] Not taking into account the effects of inelastic supply. If the net effect remaining is 5%, the amount per unit emission reduction is to be multiplied by 20.

TABLE 5.3.3 NEW ADMINISTRATIVE TRANSACTIONS REQUIRED FOR APPLICATION OF SUBSTANCE DEPOSIT TO CADMIUM[1]

CADMIUM (1986)	Thousands of tonnes	Tonnes per transaction	Thousands of transactions
Import			
-zinc ore	3800	2	2
-phosphate ore	500	0.2	2
-zinc	340	0.01	34
-refined cadmium	2300	0.1	23
-other materials	220	0.005	44
-Ni/Cd batteries	2100	0.01	210
-phosphate fert.	180	0.01	18
-other products	50	0.0005	100
Extraction			
-zinc ore	1100	4	0.3
-limestone	320	0.1	3
Export			
-zinc ore	17	0	0.5
-zinc	70	0.01	7
-refined cadmium	320	0.05	6
-other materials	850	0.005	170
-Ni/Cd batteries	300	0.005	60
-phosphate fert.	80	0.01	8
-other products	200	0.0005	400
Maximum number of transactions			**1088**

The intentional commercial use of cadmium is very complex and widespread, making the materials and products that contain cadmium even more difficult to trace. It will therefore be extremely difficult to adequately control all products, especially electronic items, plastics, and painted/pigmented products. These products should each be taxed theoretically according to their cadmium content. However, in many cases the cadmium content of an individual product is so low that the administrative process would not be worth the small deposit collected. In many of these cases, if a simple substitute for the cadmium exists, certain items could simply be banned from use and hence also import, through their general prohibition. In other cases, the items should be simply ignored, especially since their possible emissions are still controlled by the repayment part of the deposit system. In any case, the products responsible for a large share of cadmium flows, such as batteries, pigments, and certain plastics should be controlled. Some administrative difficulties will arise and some fraud will occur. It should first be kept mind that it is not the absolute level of such abuses that is relevant but the comparable level also for other types of policies with equal environmental results. Secondly, one of the startling outcomes of the study on financial instruments for the EC (Huppes et al. 1992) was that the ratio of costs to proceeds for substance deposits on sulfur, nitrogen and phosphorus is similar to that of all current duties and excises put together!

[1] Source: Huppes et al. (1992), p.159

Environmental effectiveness and costs

Environmental effectiveness works through behavioral changes. What would happen in the zinc industry, what would happen to their technologies and to their supply curves of metallic cadmium? Some extremes may define the boundaries of what may be expected. First, imagine that:

◇ zinc production remains the same
◇ the amount of cadmium in zinc is negligible
◇ no metallic cadmium is used intentionally any longer
◇ all cadmium-containing wastes will be stored in a safe site.

The deposit paid by the zinc industry, amounting to around ECU 1,500 million at current production levels, will then be fully refunded to the industry. The totally insignificant administrative costs and the costs of actual changes of behaviour are the real costs. The environmental effectiveness is 100%. Efficiency is unknown since real costs are unknown. Past use has led to "on-site" accumulations that have not yet been emitted. No deposit has been paid for these amounts of cadmium. With deposit refunding upon transformation into outflow, a large subsidy would in fact be paid for the clean-up of past sins.

Secondly, imagine that:

◇ zinc production remains the same
◇ the amount of cadmium in zinc is negligible
◇ all metallic cadmium is used in nickel-cadmium batteries, as before
◇ all cadmium-containing wastes will be stored in a safe site.

The deposit paid by the zinc industry would still be around ECU 1,500 million. Their costs of waste handling would be lower. They would have some proceeds from the sale of cadmium, and, roughly, would receive the refund back based on the now higher market price for cadmium. The domestic price of cadmium would go up roughly with the amount of the deposit, from around ECU 10 to around ECU 195.

What would happen to the battery industry, the main purchaser of cadmium? Assume that:

◇ a 95% effective deposit on batteries is introduced by industry
◇ sales are not influenced, e.g. no cadmium-free rechargeable batteries appear on the market (yet)
◇ all collected batteries are stored safely.

The price of rechargeable batteries, apart from the deposit, will be influenced, but not dramatically. It may be assumed that the battery retailers will introduce a deposit scheme at ECU 0.5 per penlite and similar amounts for other types. Their costs for collection and safe storage amount to ECU 0.06 at a current retail price of around ECU 2.00 apiece. Consumers pay the full amount for this increase in real costs at the battery level. Their costs for processing household waste and sewer would drop substantially.

These extremes will never be reached. It can hardly be imagined that the large amounts of waste from zinc ore processing, mainly consisting of such harmless minerals as iron, will be stored at relatively high costs of say ECU 2,000 per tonne (as assumed for the storage of batteries). Jarosite probably will then be used as a low grade iron ore, with the cadmium recovered in a much more concentrated form, for much cheaper storage. Metallic cadmium can be stored extremely cheaply, since in dry conditions it is one of the most chemically stable of metals. Even in wetter circumstances it is used on the strength of its extreme durability. At current prices, the use of jarosite as iron ore is not commercially attractive. If, on top of the proceeds from the process, the high costs of safe storage would be avoided, it might easily become a viable option, saving on virgin iron ore in the process.

It can hardly be imagined that technologies in the zinc industry will remain the same. The value of cadmium has been decreasing for a long time. Before the upsurge in demand for rechargeable batteries its price had gone down to less than ECU 0.50. The extraction of cadmium did not receive much attention. If the costs of storing cadmium waste could be avoided by producing more metallic cadmium, that option will be pursued as well, partly as an alternative to "jarosite as iron ore". Which of any of the large number of technical alternatives is the most attractive cannot be predicted by scientists and governments; here industrial research is a prime mover. Whatever the exact outcome will be, the real extra costs of emission reduction of cadmium will be borne to a significant degree by the users of zinc.

Assuming that:
◇ real costs of emission-reduction are extremely high, about half the deposit payments of ECU 1,000 million, that is ECU 500 million
◇ zinc users pay three-quarters of the rise in processing costs of zinc and cadmium, that is around ECU 400 million[1]
◇ zinc consumption is not influenced
◇ cadmium in zinc ore is about 0.5 percent by mass of zinc produced,
then the price of zinc will rise by ECU 0.40 per kg.

If, more realistically for the longer term, 10% of these costs will remain after technological innovations have been introduced, the price of zinc is not influenced substantially. It thus seems extremely important to introduce the deposit system (or any other effective policy for that matter) gradually but predictably. Introduction over ten years, in portions of ten percentage points each year seems a reasonable method. Full effectiveness will be reached after a decade, to increase even further as technological development is steered permanently in the direction of, now aligned, environmental and economic improvements.

Another point taken more realistically than assumed, is that a decrease in the consumption of zinc may be expected due to its price rise. Technical mechanisms are substitution and recycling. These are fundamental methods for long-term emission reduction since they reduce inflow. The remaining zinc production will lead to far less emissions and accumulations in the economy. Emission from ore processing will nearly vanish at the level of ECU 185 per kilo. Emissions will hardly occur as uncontrollable streams will nearly disappear and controllable streams, now mainly rechargeable batteries, will end in

[1] The remaining one quarter is borne by cadmium and several other materials that are co-produced.

acceptable disposal. Accumulations, both private and the "waste-to-be-processed" of firms, will lead to a loss of interest on the deposit paid and will be avoided if possible. Some "wasteful" applications of cadmium will remain, paying the full price including the deposit part. In current regulative jargon these often are called essential. Bright reds and greens, at present not exactly replaceable by other pigments, will still be used in traffic lights and in special pieces of art. The counterpart of platinum and silver jewellery may be the objects of art in the deep white gloss of metallic cadmium.

Competition

The price rise for cadmium is a general one; foreign producers pay exactly the same amount as domestic ones. For zinc, this is not the case. Real costs of domestic zinc production rise, while those of foreign producers, with a market share now of around 10%, do not rise. The formerly attractive by-product becomes a burden. Any effective environmental policy that is based on the current, or the extended polluter pays principle is bound to have this effect. Subsidizing those hurt from taxes spreads out the effect over all tax payers and, by helping to perpetuate an inefficient situation, increases the total pain. If environmental policy develops along a broader front, several industries might claim help. Apart from temporary measures to speed up reorganization, subsidizing does not make sense.

Equality

Equality is not only a characteristic of instruments but also of policies, as sets of instruments. Equality at that level is the application of one instrument to each and every decision, establishing an equal ratio between the real costs induced and the amount of emission reduced. Also taking into account transfer payments, it means applying a financial instrument to every decision, directly or indirectly if it is applied to one. The substance deposit on cadmium fully realises equality in this sense.

5.3.6 Conclusions

Cadmium is a serious environmental problem, already contaminating some essential foods to a nearly unacceptable level. The inflow of cadmium in the economy is rising. Its inflow in the global economy is in small amounts in a number of bulk ores. In some ores, it is an undesired contaminant. In zinc ore, it is valued positively and produced in combination with zinc in a single completely joint production process. Contrary to general notions, the depletion of zinc ore will not come about in the foreseeable future. Due to new applications in nickel-cadmium batteries and increased demand as a stabilizer in the steeply rising production of PVC, the price of cadmium has increased more than tenfold in the last decade.

The effective regulation of cadmium emissions is extremely difficult for three reasons. First, as an element it cannot be destroyed. Secondly, its inflow is partly price independent, as a contaminant, and partly inelastic, as a co-product in zinc production with a low share in total proceeds. Thirdly, there are many different attractive applications for cadmium.

Liability may have dubious results, with effects on concentrated local emissions only. Information on cadmium in products hardly makes sense. Information on separate collection of some items will have limited results. Emission taxes are applicable to only a limited number of flows, and would shift emissions to other flows. Product taxes cannot be applied justly, have high costs and scarcely any effect on total emissions. Direct regulations can be effective only if primarily directed at outflow processes.

The flexible response strategy consists of the substance deposit only. The level of the deposit, ECU 185 per kilogramme of cadmium, is mainly based on the incentive for separate collection, taking ECU 0.50 per penlite battery plus the costs of safe, possibly strategic, storage as a sufficient amount. At the ECU 185 level thus resulting, many cadmium applications will cease to exist. The cadmium will be immobilised and stored, in relatively concentrated forms, instead of being used, and instead of being accumulated and emitted. Use of cadmium will be restricted mainly to highly collectible applications. The current reliance on extremely costly end-of-pipe techniques, costing several thousands ECU per kilogramme of cadmium emission to air prevented, will become superfluous, at least if similar policy programmes were also set up for other the heavy metals present in wastes. A phased introduction of the substance deposit on cadmium will keep the costs of transformation low.

5.4 REGULATING SUBSTANCES: THE CASE OF NITROGEN AND PHOSPHORUS[1]

5.4.1 Introduction

Acidification and over-nutrification related problems manifest themselves throughout the whole of Northern Europe, some of the northern parts of Asia, and the northern parts of North America. They also occur regionally in many Southern countries, in all places with large populations and intensive agriculture. Examples are the Po-plain in Italy and the Adriatic. The elements concerned are sulfur, nitrogen, phosphorus and, to a much lesser extent, chlorine. Only nitrogen and phosphorus will be treated in this case study. These substances are traded extensively in products and, after their emissions, are readily transported across boundaries, both in water and air. They enter international bodies of water such as the Baltic and the North Sea. A substantial increase in these problems may be expected in the Southern countries. A limiting factor in the solution of these environmental problems at the national level is that the reduction of emissions in one country, because of their border-crossing nature, benefits other countries and common seas. A supra-national approach to these problems, therefore, could facilitate their solution. For Europe there thus seem to be good reasons to take the scope of the European Community and not that of individual countries as relevant to policy on these problems. Though limiting the analysis here to the European Community the same arguments hold, to a lesser extent, for a still higher administrative level. On the other hand, solutions as will be indicated could also be brought about by concerted action of a number of cooperating individual countries or in larger individual countries. It seems improbable, however, that in the near future environmental policies in for example Eastern and Western Europe will receive equal attention and weight in decision- making or even use the same instruments in a coordinated manner. The next sections begin with a description of the flows of the relevant nitrogen and phosphorus compounds in Europe (5.4.2), followed by an inventory of the main technical solutions (5.4.3). Then the possibilities for the macro instruments are investigated (5.4.4). The *flexible response strategy* is filled in with the most macro instruments applicable. In this case of substance policies, there again is a central role for financial instruments (5.4.5). The chapter ends with conclusions (5.4.6).

5.4.2 Substance flow analysis of nitrogen compounds and phosphorus in the European Community

Elements versus compounds

Nitrogen and phosphorus are elements that play an essential role in all processes related to life, both beneficial and harmful. Their role in organic and inorganic chemistry is rather limited. In animal husbandry and agricultural production they function together at a cellular level. Measures related to the one substance therefore often relate to both substances together. That is the practical reason for treating them here as one case. Should the analysis extend to all compounds encountered or is a restriction necessary? This question relates quite directly to the kernel of the substance approach in environmental policy. If the element is chosen, including any of its compounds, there is a very limited number of main flows. These flows, however, will often be diverse

[1] This chapter is to a large extent based on Huppes et al 1992; van der Voet 1992; and, for the Dutch situation, on van der Voet, Witmond and Huppes 1989.

chemically. Consequently, each compound may be related to different types of environmental problems, in quantitatively differing ways. If, on the other hand, only one compound is chosen a very large number of flow charts must be constructed, for each compound, that are also not independent. Changes in one flow, such as oxidation of NH_3, are an outflow in the NH_3 system but at the same time they constitute an inflow in the NO_3 system. If compounds are grouped at a certain level, what should be the criteria? The most practical would be a chemical criterion, with limited transformations between the groups chosen, based on the shared contribution to several problems of the compounds within the group.

For nitrogen, what could be the definition of the nitrogen compounds relevant for instrument design here? The ready transformation between different N compounds means grouping them is not feasible. There is only one clear cut subdivision, that between the quite non-reactive nitrogen molecule, N_2 and all other compounds. The nitrogen molecule, being stable and unrelated to any environmental problem, is disregarded here. For practical reasons, certain other stable compounds of nitrogen, such as polyamide, have been disregarded as well. Again for practical reasons, nitrous oxide (N_2O, or laughing gas), which may make a substantial contribution to global warming, is omitted from explicit analysis as well. Nitrate leaching into ground water exceeds norms for drinking water in many countries. That problem is not treated in the analysis directly. All compounds related to the biological nitrogen cycle are included, explicitly or implicitly. The nitrogen analysis thus structured relates to both the over-nutrification and acidification problems, but not so much to the problems of global warming and toxicity. There is no depletion problem of nitrogen, as N_2 amounts in the air are very large in comparison to its use, and the use of nitrogen containing resources, such as coal, adds to this large stock in the environment. The two main types of nitrogen compounds, NO_x and NH_x, cause over-nutrification directly. Acidification is caused directly by NO_x and indirectly by NH_x, through its transformation in the soil into NO_x by microbial activity. In the economy, and also the environment, nitrogen compounds may be formed chemically, its main economic inflow, and may be broken down back into the harmless nitrogen molecule, N_2, through denitrification, generally after first being emitted into the environment.

For phosphorus, all compounds can be included in principle, with phosphates, acids of phosphorus and lipids being primary examples. Some specific compounds related to toxicity problems, as in phosphorus containing pesticides, have been omitted from explicit analysis for practical reasons. Phosphorus is one main cause of over-nutrification. In arable soils, this problem arises when soils become saturated with phosphate. Additional phosphate used on that soil will then become mobile and find its way to ground and surface water. The amounts of phosphate available in the lithosphere and seas do not indicate a depletion problem that is as serious as over-nutrification and acidification. The reserves of the most concentrated phosphate ores are limited of course, by definition. At higher price levels, phosphorus does not seem to have a limited availability. As long as enough energy is available (and there is, see the next case study), phosphates can be produced in the desired forms and concentrations. The depletion problem is therefore disregarded. Thus the phosphorus analysis is restricted to the over-nutrification problem. Phosphorus, being an element, cannot be formed and broken down chemically.

The central aim of this analysis thus becomes the prevention and reduction of over-nutrification and acidification related to nitrogen in all its compounds with other elements and to phosphorus in all its compounds. Limiting the range of the analysis in this way does not imply that other substances might not contribute to the problems of over-nutrification and acidification. Potassium is another potentially over-nutrificating substance and sulfur is the main acidifying element. Nor does it imply that the two substances might not be related to quite different environmental problems, as has been indicated. The limits chosen are arbitrary in a way; climate effects could have been included by another choice of compounds, as would have been the case with potential toxic effects. The flow analysis would then become much more complicated since only relatively minor sub-flows would cause the main contributions to these other types of environmental problems. The instruments to be developed would have to differentiate between the contributions to each of these problems, requiring a substance flow at the more desaggregated level of individual compounds. If the overall flows of nitrogen compounds and phosphorus are reduced, for example through the flexible response strategy, there will also be a reduction in their contribution to other problems, such as the health effects of nitrites and the global warming effects of nitrous oxide. Intentionally created compounds, such as pesticides, will not co-vary.

The substance flow analysis is given for each of the two groups of compounds separately. The results are based first on research in the study for the European Commission mentioned. These data have been updated and detailed, see van der Voet et al. (1993b, in prep.).

Nitrogen: substance flows and problems caused
Nitrogen is treated here as jointly contributing to the environmental problems of acidification and over-nutrification, through the ready transformation of the main groups into each other in agriculture and the environment.

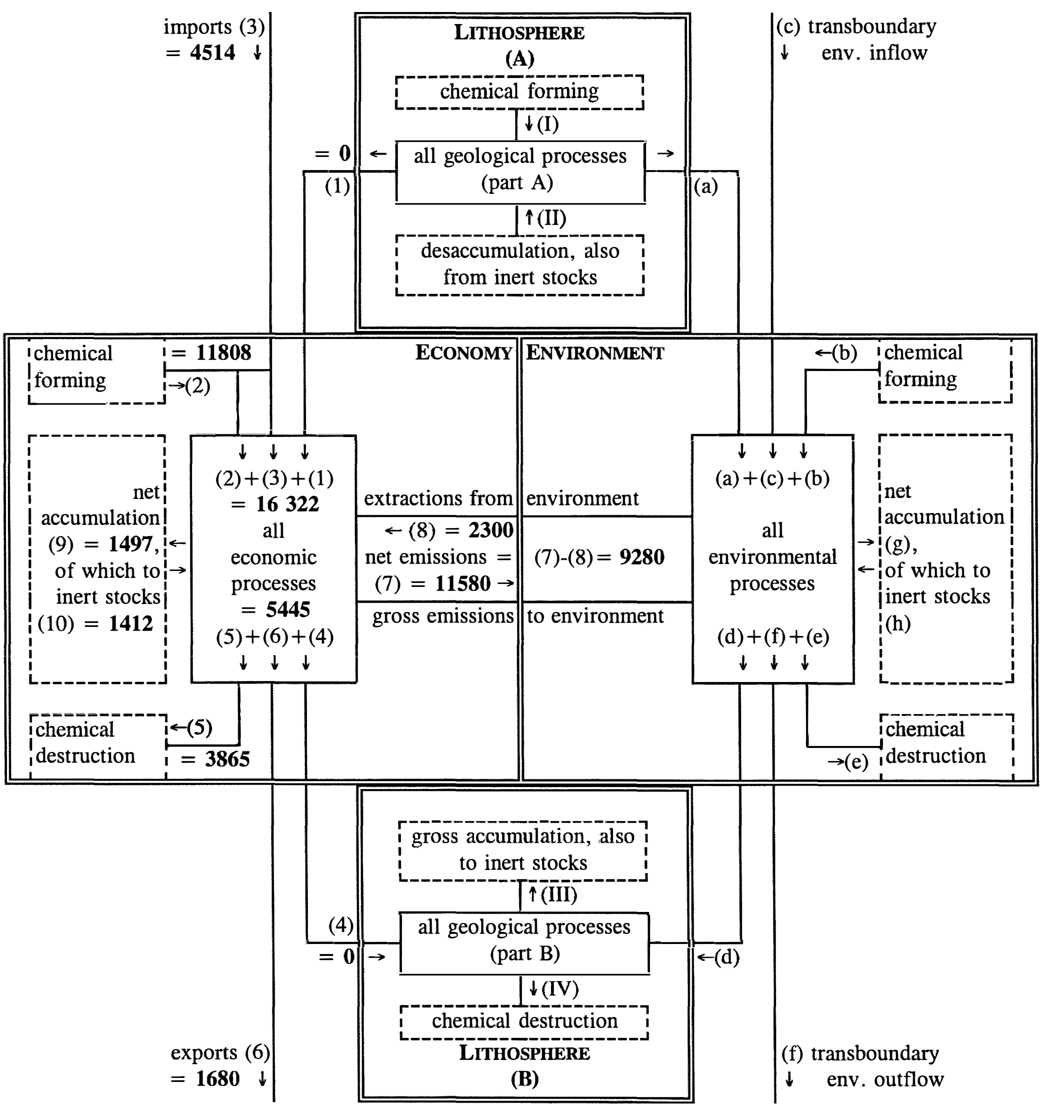
NH_x emissions are almost without exception related to agriculture, the main exception being the nitrogen taken out of oil in refineries in reduced form. NO_x emissions have various sources, of which traffic and large furnaces such as those used in power plants and certain factories are the most important. The origin of N deposition may be described in terms of direct causes as emissions, in terms of the economic activities directly responsible for these emissions, and in relation to the final origin ("borders" of the substance flow diagram), where N compounds enter the economy.

The over-nutrification problem caused by N-compounds can almost entirely be traced back to economic activities in agriculture. As much as 90% of the soil load of nitrogen is due to fertilizing, and only 8% to atmospheric deposition. The agricultural origin of the soil load thus totals about 94%, since 58% of the atmospheric deposition also can be traced back to agriculture. As to final origins, there is a major role of ammonia production from N_2. For ground water pollution and also for the pollution of surface water, waste water processing and agriculture are the major direct sources. Waste water is at the end of the food chain that starts in the production of artificial fertilizer and in agriculture. These emissions are ultimately based nearly exclusively on the production of fertilizer. The acidification problem, defined as one of acid precipitation, is related first to NO_x but also to ammonia. The acidifying effect of fertilizer application is disregarded.

NO_x is mainly formed in the burning of fossil fuels. Some is removed from the combustion gases, in denox installations, the remaining part is emitted directly.

There are thus two fully independent types of nitrogen flows. The one, described above, is related to organic processes, starting at the mainly industrial production of fertilizer and

FIGURE 5.4.1 GENERAL SUBSTANCE FLOW SCHEME OF FLOWS AND ACCUMULATIONS OF NON-N_2 NITROGEN IN THE ECONOMY OF THE EC, 1987, IN THOUSANDS OF TONNES (SOURCE: VAN DER VOET ET AL. 1993B IN PREP., PRELIMINARY DATA.)



passing through agricultural processes, with both NO_x and NH_x involved in economic flows. The other is related to combustion processes, with a flow of only NO_x through the economy, and is directly emitted when formed. In the environment, this NO_x may also be transformed into NH_x. Policy instruments for these two independent flows might differ, if there are no boundary problems in their application.

See for nitrogen compounds figure 5.4.1 (above) for a general survey of flows and accumulations in the economy, and table 5.4.1 for a somewhat more detailed summary of these, and analogously for phosphorus figure 5.4.2 and table 5.4.2.

TABLE 5.4.1 FLOWS AND ACCUMULATIONS OF NUTRIFICATING AND ACIDIFYING NITROGEN COMPOUNDS IN THE ECONOMY OF THE EC, IN 1987, IN THOUSANDS OF TONNES. (SOURCE: VAN DER VOET ET AL 1993B IN PREP., PRELIMINARY DATA.)

1[1]. Extraction from lithosphere		4. Deliverance to lithosphere	
Fossil resource extraction	pm	None	
Subtotal from lithosphere	*pm*	*Subtotal to lithosphere*	*0*
Subtotal to lithosphere	0		
Net inflow from lithosphere	**pm**		
2. Chemical forming in the economy		5. Chemical destruction in the economy	
NOx from N_2 combustion	3110	denitrification in sewage treatment	263
industrial N_2 fixation[a]	8398	denitrif. in central manure treatment[a]	0
agricultural N_2 fixation	300	denitrification in agricultural soil	3602
Subtotal chemical forming	*11808*	*Subtotal chemical destruction*	*3865*
Subtotal chemical destruction	3865		
Net chemical forming	**7943**		
3. Imports		6. Exports	
fossil fuels	pm	wood	0
other N-containing products	pm	manure based fertilizer	0
NH/NO containing chemicals	1295	processed foods[a]	227
fertilizers	1541	other agrarian products	504
wood[a]	13	fertilizer	485
agrarian consumer products	55	NH/NO containing chemicals	464
food to be processed	268	fossil fuels	pm
fodder	1342		
Subtotal imports	*4514*	*Subtotal exports*	*1680*
Subtotal exports	1680		
Net imports	**2834**		
Total structural inflow	*16322*	*Total structural outflow*	*5445*
Total structural outflow	5445		
Net structural inflow[2]	**10877**		

[1] Numbers correspond to those in the flow scheme.

[2] Net structural inflow = total structural inflow minus total structural outflow = [(1) + (2) + (3)] - [(4) + (5) + (6)]

8. Inflow from the environment		7. Outflow to the environment		
fish as fodder	pm			
fish for human consumption	266	**To ground water**		**3602**
dredged sediment	pm	from agricultural soils[a]	3602	
wood	422			
deposition on agricultural soils	1312	**To inland waters**		**2386**
		sewage treatment[a]	1806	
		central manure treatment	0	
		fodder industry[a]	171	
		food industry	175	
		vegetable production	234	
		To seas		**20**
		sludge (from inland extraction)	pm	
		sewage treatment	20	
		To air		**5572**
		waste treatment	0	
		sewage treatment	0	
		central manure treatment	0	
		vegetable production	1535	
		animal production	771	
		NH3 industry	58	
		combustion processes[a]	3208	
		To soil		**0**
		None		
Subtotal from environment	*2300*	*Subtotal to environment*		*11580*
		Subtotal from environment		2300
		Net emissions		**9280**
Total inflow into the economy		Total outflow out of the economy		
*extraction from lithosphere	pm	*to lithosphere		0
*chemical forming	11808	*chemical destruction		3865
*imports	4514	*exports		1680
*extraction from environment	2300	*emissions to environment		11580
Total inflow	*18622*	*Total outflow*		*17125*
Total outflow	17125			
Accumulation on balance	**1497**			
9. Accumulation in the economy *from balance:*	*1497*			
In mobile stocks				
products[a]	84			
In inert stocks (10)				
landfill[a]	1412			
Subtotal accumulation	*1496*			
Rounding errors[a]	1			
Total accumulation	**1497**			

[a] is balancing item.

FIGURE 5.4.2 INFLOWS, OUTFLOWS, NET EMISSIONS, AND ACCUMULATIONS OF PHOSPHORUS IN THE EC, 1987, IN TONNES; GENERAL SUBSTANCE FLOW SCHEME

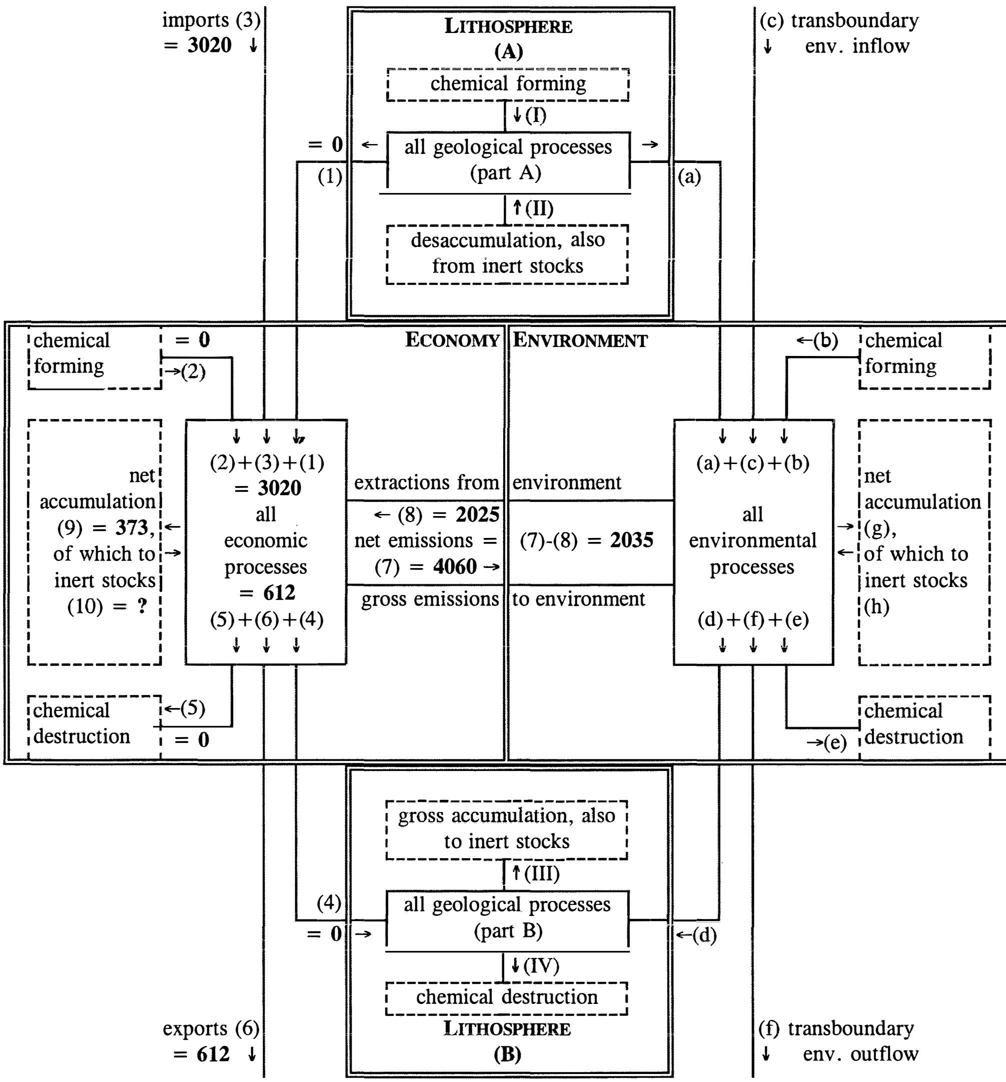


TABLE 5.4.2 FLOWS AND ACCUMULATIONS OF PHOSPHORUS IN THE ECONOMY OF THE EC, 1987, IN TONNES. SOURCE: HUPPES ET AL. 1992

1[1]. Extraction from lithosphere		4. Deliverance to lithosphere	
None		None	
Subtotal from lithosphere	*0*	*Subtotal to lithosphere*	*0*
Subtotal to lithosphere	0		
Net inflow from lithosphere	**0**		
2. Chemical forming in the economy		5. Chemical destruction in the economy	
None		None	
Subtotal	*0*	*Subtotal chemical destruction*	*0*
Subtotal chemical destruction	0		
Net chemical forming	**0**		
3. Imports		6. Exports	
P ore	2079		
P fertiliser (net)	557	P fertiliser	(netted with imports)
P containing materials/products	279	P containing materials/products	504
food products	85	food products	108
fodders	20		
Subtotal imports	*3020*	*Subtotal exports*	*612*
Subtotal exports	612		
Net imports	**2408**		
Total structural inflow	*3020*	*Total structural outflow*	*612*
Total structural outflow	612		
Net structural inflow[2]	**2408**		
8. Inflow from the environment		7. Outflow to the environment	
fish for human consumption	38	fertiliser	1951
dredged sediment	pm	manure	1431
wood	60	sewage sludge to seas	3
food	406	sewage sludge to soil	71
grass	1166	sewage effluent	504
fodder	355	industrial effluent	100
Subtotal from environment	*2025*	*Subtotal to environment*	*4060*
		Subtotal from environment	2025
		Net emissions	**2035**
Total inflow into the economy		Total outflow out of the economy	
*extraction from lithosphere	0	*to lithosphere	0
*chemical forming	0	*chemical destruction	0
*imports	3020	*exports	612
*extraction from environment	2025	*emission to environment	4060
Total inflow	*5045*	*Total outflow*	*4672*
Total outflow	4672		
Accumulation on balance	**373**		
9. Accumulation in the economy			
not specified[a]	313		
wood	60		
Total accumulation	**373**		

[a] is balancing item.

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[1] Numbers correspond to those in the flow scheme.

[2] Net structural inflow = total structural inflow minus total structural outflow = [(1) + (2) + (3)] - [(4) + (5) + (6)]

Phosphorus, substance flows and problems caused
Like nitrogen, phosphorus contributes to the over-nutrification of soils and surface water. Although phosphoric acid is used in some quantities there is no relevant contribution to acid deposition. Toxic effects of some phosphorus compounds are not considered here specifically. All flows of the element phosphorus are taken into account. Virtually all phosphorus enters the economy from the lithosphere, mainly indirectly, through the import of ores and processed ores. This was also the case with cadmium, but not with nitrogen compounds that enter through chemical formation and, as the element nitrogen, come from the atmosphere. There is no outflow to the lithosphere, neither directly nor indirectly through the very limited exports. Chemical destruction of an element is not possible. Accumulation is very limited. Thus, virtually all structural inflow is transformed into emissions, the structural outflow being very low. See the general flow scheme on the preceding page and the table below for more details.

Summary on flows and accumulations
Nitrogen (non-N_2 compounds) primarily enter the economy directly through chemical formation 63%, and indirectly through imports, 24%. Chemical formation for agricultural applications is intentional. About 17% percent points of the 63% is not intentionally formed, in combustion processes. It leaves the economy through emissions to the environment which account for over 70% of outflow, the other flows are exports, leading mainly to emissions abroad, and chemical destruction. There are no flows to the lithosphere. Accumulation in the economy is nearly absent.

Phosphorus almost exclusively enters the economy directly through the extraction from substrate, or indirectly through imports, very similar to cadmium and unlike nitrogen. Almost all leaves the economy as emissions. Exports and a return to lithosphere are nearly absent. Accumulation in the economy is minimal as well, at 7%, quite the opposite from the situation of cadmium.

5.4.3. Technical solutions
Technical solutions to the over-nutrification and acidification problems caused by N compounds and P compounds may be formulated at several levels. Since the supply of the basic inflows is quite elastic, there is not much divergence between micro improvements within the economy and the macro effects at its boundary.

First, there is a number of end-of-pipe solutions. Waste water treatment may be improved to diminish the nutrient load on surface waters and seas. Car exhaust gasses may be cleaned by catalytic convertors. NO_x from large furnaces may be removed in denox installations. Manure storage may be closed or treated chemically to prevent the emission of ammonia. The ventilation air of farms may be cleaned, etc. If effective, these measures ultimately will lead to either a reduced inflow, especially with phosphorus, or an increased outflow, especially with nitrogen, through denitrification.
Secondly, on the basis of a more complicated analysis, process integrated improvements may be designed. Fodder-meat conversion ratio's may be improved in animal husbandry. Mineral uptake, also in animal husbandry, may become much more efficient through the addition of specific enzymes to the fodder, themselves a co-product of specialized processing of manure. Manure might also be processed into a fertilizer that is comparable to the artificial fertilizer now used in such large amounts, etc. Fertilisers may be applied

at higher efficiencies. Combustion processes may be changed to reduce NO_x forming. Some coal gasification techniques, e.g., do not produce NO_x at all.
Thirdly, improvements may be realized by changes in the products used. Combustion engines in cars may be replaced by electromotors using electricity from NO_x free sources. Trains, boats and bicycles may replace cars and planes. Meat may be replaced by vegetables. Nuclear power, wind energy and solar power may replace fossil fuels. Phosphoric acid in industrial applications may be replaced by organic acids, etc. Of course, we also might consume less of everything related to nitrogen and phosphorus, by shifting consumption to other items.
Next, the economic inflow might be diminished, and the outflow increased. In particular the import of phosphorus ore might be diminished, as well as the production of nitrate fertilizer. For nitrous oxides, usually emitted when formed, emission reduction and inflow reduction are the same. At the outflow side there is a similar direct relation with emission reduction in waste water treatment. Nitrogen compounds may be reduced in these processes to the harmless nitrogen gas.
Fifthly, recycling, especially the improvement of already existing recycling processes, is one of the most important options in limiting inflow and emissions, for both nitrogen and phosphorus. The flows from animal husbandry in particular are first stored inadequately, with large losses of ammonia, and then applied in such an inefficient manner that large losses to the environment again occur. Several technologies are available to treat manure in a way that leads to improved recycling and acceptable disposal. Most of them are not viable commercially.
One special option is remarkable, that of the combined treatment of the agricultural and the industrial combustion waste flows. The NO_x from fuel combustion at power plants might be combined with the ammonia from a nearby manure processing plant using the waste heat of power generation. Together these flows could form N_2 and water, or fertiliser.

What is common to most of the options indicated is not the application of one technology to reduce emissions but the paramount importance of daily behavioral choices, which require both knowledge and attention.

5.4.4 Policy instruments

The analysis here will specify the instruments applicable to several streams. The combination of some of them into a more or less consistent flexible response strategy is the subject of the next section. The strategy developed will involve a certain amount of quantification. Which instruments can be applied to the different flows of nitrogen and phosphorus? Stepping down the ladder of freedom and efficiency, the following possibilities can be identified:

Structural	* (Extended liability)
Cultural	* Standard methodology for LCA
Financial	* Substance deposit
	* Uniform emission tax
	* Estimated emission tax
Prohibiting	* General emission design standard
	* Estimated product standard, LCA
	* Estimated emission design standard

Structural instruments

First, the general principles of property, especially the liability rules protecting it, might be applied. Strict liability is required for a reasonable chance of application. With N compounds, nutrification and acidifying effects could be apportioned to the emitters only in very exceptional cases. Prominent examples are the damage to nature areas in the direct vicinity of single sources or a small number of sources, as in the case of intensive husbandry and waste water treatment installations. The effects of such rules would not so much regulate total emissions as the location of certain installations. With P compounds applicability is still more limited because of their less volatile nature and slower movement. Liability rules will not contribute much to the emission reduction of nitrogen and phosphorus.

Cultural instruments

The life cycle analysis of products, directed at all supra local problems, would also take into account the effects of nitrogen and phosphorus on acidification and over-nutrification. Overall effects of LCA are limited. This applies on average to each individual substance. Focusing LCA on nitrogen and phosphorus seems highly undesirable. Setting up a partial informational instrument would seriously jeopardize the completeness of the information that might be provided through the LCA-based product information.

Some second-choice informational instrument might be useful. In well planned technological development, aims are several times set against the costs, a procedure that squeezes out any costs that do not contribute to the aims. If the market does not pay the extra costs for emission reductions of N and P the technology will not be developed, apart maybe from subsidies. Sometimes, pollution prevention pays. In that case firms may introduce economic improvements while at the same time reducing emissions. In the Netherlands, a software program for farmers has been developed by CLM Utrecht that may have such a doubly positive effect. It helps set up the firm's input-output analysis of nitrogen and phosphorus. One should be cautious in attributing all future improvements in nutrient efficiency to this informational instrument, for two reasons. First, current regulations in agricultural markets in the EC favour any technical improvement that reduces costs, since for many products an increase in production now reduces off-farm prices drastically. Cost reduction generally may be realised only if the use of inputs is diminished by higher efficiency and thus fewer accompanying losses through emissions. Secondly, a more general reason for limited effectiveness is that only technological followers, relying quite passively on others for guiding their business development, may be influenced. Technological leaders had already set up a nutrient analysis long before. This means that the time lag between technological leaders and followers may be reduced by the active supply of information. After the initial positive effect, a constant policy effort is required to perpetuate this situation. If the effort stops, the time lag in technological improvements will return to normal and no effect will remain. All the same, this second-choice informational instrument may have some effects, especially in agriculture.

Financial instruments

The substance deposit is applicable to all main nitrogen flows related to agriculture and to virtually all phosphorus flows. The emission tax is applicable to some of these flows, where direct emission measurement is possible. Waste water purification and industrial processing of agricultural products are the primary examples. There are no instances where the emission tax is applicable and the deposit system is not. Thus, in the choice between them, the deposit system should be used. The emission tax is not then to be applied to waste water purification and the industrial processing of agricultural products since the deposit system applies to them. Only denitrification into the harmless N_2 forms the basis for deposit refunding. It may be noted that in no instance does the implementation of the deposit system directly touch the chief regulatees, the millions of farmers of the Community. Producers and importers of fertilisers and importers of fodder pay the deposit, while exporters and specialised waste processors receive the refund of the deposit. Implementation is not worked out here in detail, see Huppes et al. 1992 for these details. It is very similar to that of cadmium, with the national customs and excise offices being the existing organization responsible for both deposit payments and refunding for the Community-wide scheme. The second main inflow of nitrogen is that of chemical forming in combustion engines. Here the deposit system is not applicable at all. The emission tax is applicable only to large furnaces.

A nearly infinite number of second-choice financial instruments is available since any behaviour and any product may be influenced by taxing or subsidizing it or its alternatives. However, the aim here is to find financial instruments that come as near as possible to the ideal emission tax, applied at a level as encompassing as possible. This search is especially relevant where the substance deposit and the emission tax are not applicable. This seems the typical situation with small-scale combustion and incineration processes, where small is therefore not beautiful. Small furnaces, stoves, central heating systems are the main stationary sources, and non-electric transport are the mobile sources. The Motor Vehicles Group, a body advising the European Commission on ways to reduce global warming gasses from cars, has discussed several options for such second-choice instruments. The degree to which they are sub-optimal is closely related to the side-effects occurring. The British representatives propose a tax based on falling behind on a norm for increasing fuel efficiency, to be paid out to those exceeding it. The Germans propose a tax based on limits of expected emissions, differentiated according to weight or engine size. Such taxes are related to emissions, but only indirectly since it is factors other than actual emissions that are taken into account, such as "car size" and "possibilities for increased fuel efficiency". Discussions then centre on questions as to which producer would benefit, relatively, from which scheme, with positions taken related to the perceived commercial interest[1]. The efficiency in reaching environmental improvements is limited because factors other than weight or fuel efficiency are disregarded and are quite decisive. The main factor not influenced by either scheme is the number of kilometres driven per car. A second major factor not taken into account is the amount of emissions per kilometre. With NO_x (but not CO_2 for example), the principal technical factors not influenced by the schemes proposed are the amounts first produced in the engine and then reduced by analytic conversion. For NO_x (again in contrast to CO_2) no clear indicator of car emissions is available. It is possible, however, to arrive at a tax scheme that comes much closer to the emission tax. It is possible to test a car at the regular mandatory checks, now introduced in most EC countries, on the amount of NO_x

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[1] Interestingly, the reasoning in such cases seems so muddled that the British advocate the efficiency scheme because the German cars use such a relatively large amount of fuel, see the Economist of 18-1-1992, compared to. It is to be expected, however, that high current gasoline consumption gives the best opportunities for improvements.

in its exhaust and tax the car accordingly. This would still omit the distance driven. The number of kilometres may be measured quite easily as well and may be included in the taxing scheme. Then the only factors not influenced by the tax are the speed and style of driving, and the maintenance of the motor between the regular checks.

For small stationary sources no controls are available to which to anchor the financial instrument. The easiest alternative would be a tax on each such appliance sold, based on the expected amount of NO_x emitted during its working life. This would not influence the way a burner is maintained and used. It would induce people to buy improved appliances, thus stimulating technological improvements in a way fully equivalent to the fictitious emission tax. It would be a substantial improvement compared to the static effects of prohibiting instruments used now.

Prohibiting instruments
Real emission measurement is practically impossible at many sources, e.g. small furnaces and in mobile combustion. General design emission standards then are not applicable, only the estimated variant. With car exhausts, for example, estimated emission design standards have been developed in most countries, including standards for NO_x. Such standards can force the worst emitters off the market but usually not the average offender[1].
There is a very important difference between a dynamic prohibiting instrument such as the emission design standard and the technology-determining technical standard. For NO_x emissions from cars, this is the difference between exhaust standards and the obligatory catalytic converter. The choice for catalytic converters might at first glance seem an extremely sensible option for diminishing emissions. And, without doubt, it is a measure with generally undisputed environmental advantages. It may be introduced by prohibiting cars without such converters or, as in the Netherlands, by nearly equivalent second-choice financial instruments that make the converter-less car more expensive. Further consideration, however, that takes into account the societal macro-surroundings of such a micro-measure poses questions might as to the dynamics of technological development. The long-term net effect of such a use of standards might well be negative, because of two mechanisms, an economic and a technical one. The pushing of slower rivals into these newer technologies will diminish the development premium for the active developers. This is the economic mechanism that may slow down technological development and may thus reverse long- term net effects. The other mechanism is directly technical. Developments in motor technology indicated the high potential of the High Compression Lean Burn ("HCLB") motor, both in reducing NO_x emissions and for lowering fuel consumption. This engine might, if fully developed, be superior to a future engine with catalytic convertor. The development towards this HCLB alternative has probably been halted by the choice for the catalytic convertor, which requires another type of engine. It is difficult to avoid such negative effects on what are potentially much

[1] Technologically imposed estimated emission design standards have been applied successfully, especially for cars in the US.

TABLE 5.4.3 THE APPLICABILITY OF SEVERAL INSTRUMENTS TO DIFFERENT FLOWS OF NITROGEN AND PHOSPHORUS

flows of N and P to be regulated	structural instruments	cultural instruments	financial instruments	second choice financial instruments	direct regulations
animal husbandry	-- only situational	- incompat. with prod. information	++ substance deposit only	+/- fodder taxes	- no emission measurem., limited
vegetable production	-- only situational	- incompat. with prod. information	++ substance deposit only	++ taxes on fertilizer	- no emission measurem., limited
sewer water purification	+/- sludge and ashes only	-	++ both emiss. tax and subst. dep.	?	++ quality and emission standards
industrial emissions to water	-- only situational	-	++ both emiss. tax and subst. dep.	?	+ effluent norms
large furnaces	+/- sludge and ashes only	-	++ emission tax	?	++ emission standards
small furnaces	--	-	+/- emission tax	+/- tax on estimated emissions	+/- quality standards
mobile combustion	--	-	--	++ tax on estimated emissions	+ product quality standards
recycling	+/- works against recycling	+/-	++ mainly subst. deposit	- product deposit	+/- limited

more attractive alternatives. And we will never know what exactly we missed because the development has been cut off and will never materialize. It is even more difficult with direct regulations to set in motion the mechanisms and motives for continued technological environmental improvements.

5.4.5 *The flexible response strategy: a mix*

Extended liability can hardly play a role in general emission reductions for N and P. The role of life cycle analysis remains limited as well. The impetus for far reaching emission reductions comes from financial instruments. The substance deposit applies similarly to the agricultural flows of N and P.

All flows of phosphorus may be regulated with the substance deposit, including those from industry. All nitrogen flows related to agriculture may be brought under the deposit system as well. The deposit system may be applied to relatively minor industrial applications of nitrogen, e.g. in ammonia and organic chemistry. Although the emissions from sewerage purification and the industrial emissions related to agriculture could be regulated by emission taxes this option should not be used. Such taxes, added to the substance deposit, would lead to double the taxation of these flows.

All nitrogen emissions from large-scale combustion processes and the emissions to water of the non-agricultural chemical industry can be taxed per unit. The other main type of nitrogen emissions results from combustion processes. Emissions from large, stationary sources such as factories and large power plants also are easily covered by an emission tax.

More difficult to control are the emissions from small stationary sources: small heating and power installations, stoves, stationary engines and the like. The estimated emission tax could then be applied on a yearly basis for example. The lack of existing control mechanisms makes implementation difficult. The next best choice is a product tax on the expected nitrogen emissions during the life span of the apparatus, applied at the purchase of the installation.

For emissions of mobile sources, emission monitoring for taxing purposes is difficult, and so is the application of an emission tax. A substance deposit system, however, is equally difficult to apply. To a large extent the emissions result from the transformation of N_2 taken from the air into NO_x. This inflow is as difficult to monitor as the emissions. The most preferred second-choice financial instrument is the estimated emission tax. It can be applied to cars through actual emission measurements at yearly technical checks, combined with data on the distance travelled. For trains, boats and planes there are administrative controls already, with different aims. Implementation in these cases would not be difficult.

Implementation
The flexible response strategy may be initiated at the central level of the Community, i.e. the option assumed here. In principle it also could start through the concerted action of the member states. Including third states into the system is fairly easy. The implementation of both the deposit and the emission taxes is a task primarily for the national Customs and Excise Offices of the member states. The criteria for repayment of the deposit are set by the department of the environment, based on an EC directive. One of the lessons to learn from the common agricultural policy is that member states will not give high priority to costly administrative activities, that work as a disadvantage to their firms, in the form of extra payments or fewer refunds, while the extra money brought in, or the refunds saved go to a central account of the Community. When administrative priorities are set such activities related to implementation as control and enforcement will be primarily for those activities that bring in money, other things being equal. It seems wiser therefore to leave the proceeds of deposit and taxes at the national level and refund also at the national level only. The slight distributional injustice that some countries import more or export more than others must then be accepted. Payments and refunds should be made in the countries where these processes actually occur.

Tax amount and effectiveness
The level of the deposit, the emission tax and the estimated emission tax should all be equal. Every polluter then pays an equal amount for an equal emission into the environment, which is both just and efficient. On the basis of the critical tax levels given

in table 5.4.4 and the substance flows given, the level of tax could be set to lead to "substantial reductions". An important technique is the central manure treatment, without which the agriculture related emissions cannot be reduced enough, at least in areas with a dense cattle population. If this technique is to become commercially attractive, the tax amount should be around ECU 1.20 per kg of nitrogen and per kg phosphorus. Incidentally, this is not much higher than the tax on, inter alia, N emissions to water that has very successfully been levied in the Netherlands on industrial emissions to water for over two decades now, see Huppes and Kagan (1989). Putting the same amount on both N and P is a debatable choice, since phosphorus does not contribute to acidification at all. However, phosphorus cannot be broken down chemically, as is the case with nitrogen compounds. How much one kilogramme of phosphorus contributes to over-nutrification compared to one kilogramme of nitrogen is hard to tell. The curious effect is that if there is too much of the one substance, the other becomes the limiting factor on plant growth. Its extra emission than causes the extra growth, because its emission is relatively low!

In the study on financial instruments for the European Commission the deposit and the taxes were set at ECU 1.20 per kilogram. Based on current techniques and on current prices, a comparative static analysis indicated a large reduction of net emissions to the EC

TABLE 5.4.4 CRITICAL EMISSION TAX LEVEL FOR VARIOUS TECHNIQUES FOR REDUCING N AND P EMISSIONS

technique/measure	amount of tax (ECU/kg N or P)
careful application of fertilizer	ca 0
low NO_x techniques gas engines[1]	0.04
IFNR/low NO_x techniques large furnaces[1]	0.18
sewage effluent denitrification[2]	0.31
covering manure after spreading[2]	0.36
slurry injection[2]	0.36
fodder adjustments[2]	0.44
IFNR/low NO_x techniques (most categories)[1]	< 0.99
NSCR large stationary sources (most categories)[1]	< 1.06
central manure treatment[2]	1.19
"clean engine" traffic[1]	1.65
SCR large stationary sources[2]	6.40
P free detergents[1]	11.50
3-way catalyst motorcars[1]	13.21
stable ventilation filters, poultry[2]	14.92
stable ventilation filters, pigs[2]	32.36

[1]From: Jantzen 1990
[2]From: Huppes et al. 1992

environment of 30 percent. The techniques used indicate a reduction in emissions to air, i.e. most of the acidifying emissions. Reductions in over-nutrification will be comparatively lower. This improvement would be realised in about ten years time, after the depreciation of existing installations. The dynamics of technological change that start with financial instruments have not yet been taken into account since these developments are unknown. If assumed to be similar to the average technological development in the market, with a cumulative cost reduction each year of around five percent, the dynamic effect would in the not-too-distant future be larger than the comparable static effect, halving emissions per unit of product every 12 years.

Equality

Equality not only is a characteristic of instruments but also of policies, as sets of instruments. Equality at that level is the application of one instrument to each and every decision, establishing an equal ratio between the real costs induced and the amount of emission reduced. Taking into account transfer payments as well, it means applying a financial instrument to any decision, directly or indirectly if it is applied to one. The mix of instruments comprising the flexible response strategy is restricted to financial instruments that, together cover all flows only once. It thus fully realises equality at the level of the instrument set.

5.4.6 Conclusions

The inflow of phosphorus into the economy is nearly exclusively from phosphate ore. With nitrogen there is one cluster originating from N fertilizer production and one cluster from combustion processes. Fertilizer production of N and P is not inelastic, as was the case with cadmium. Therefore, measures that make sense at the micro level here are usually effective at the macro level as well, though to a lesser extent.

The flexible response strategy for nitrogen and phosphorus consist mainly of financial instruments, with a limited role for extended liability and life cycle analysis. The main elements are:

◇ *Substance deposit* for all nitrogen flows directly or indirectly related to agriculture and for all phosphorus flows.
◇ *Emission tax* for large stationary NO_x sources
◇ *Estimated emission tax* for some small and all mobile installations, the latter based on yearly emission measurement and distance driven
◇ *Estimated emission tax* for all other combustion appliances, based on expected life span emissions

The implementation of both the substance deposit and the taxes is primarily a job for the Customs and Excise Offices of the member states. Proceeds and payments should *not* be cleared between member states of the EC, in order to create the right implementation incentives.

The level of the substance deposits and the (estimated) emission taxes is to be set at around ECU 1.20 to reach emission reductions on the order of thirty to sixty percent in the next decade.

5.5 ENERGY DEPLETION AND GLOBAL WARMING[1]

5.5.1 Introduction

In Europe there is a long tradition in taxing energy. In Japan, a country virtually without energy resources, taxes were raised in the Seventies, immediately after the first price explosion of crude oil in 1973. There is clear evidence that substantial energy taxes lead to a substantially smaller energy consumption per unit of GNP than in countries where such taxes are lacking. In Japan a litre of gasoline costs nearly four times as much as in the US. In Europe there are differences between countries but the general level is comparable to that of Japan. In the US the energy consumption per ECU of national income is about twice as high as in Japan and Western Europe. In the former Soviet Union the price of energy was about half the level of world market prices, through deliberate policy. The energy consumption per ECU of GNP there is again about twice as high as in the USA. Thus, in Western Europe and Japan, energy depletion and CO2 emissions from energy use, per ECU of GNP, are half that of the US and about a quarter of that of the former Soviet Union. See Knook (1991)[2] for a survey on the relation between energy use and national product.

After the tenfold price rise of oil in the mid-Seventies, pre-tax market prices have been declining since 1981. In real terms they have been more than halved. Since the end of First World War, prices have been steadily declining, roughly, see figure 5.5.1., with the price explosion of the Seventies the only exception. No structural change to reverse this declining tendency has occurred. Since the Seventies a number of incidents have halted the decline. First, there was the war between Iraq and Iran that halted part of their oil exports. Then the Iraq-Kuwait conflict kept the second and third largest producers temporarily off the market, with Iraq now threatening to return to export production. The sources of energy now have become more diversified than ever before. There is no reason to assume a deviation from the long term trend for the decades to come. In 1990, prices now nearly returned to their long term trend value (assuming a price of $18 per barrel and an inflation of 2% a year). Decreasing costs of production, through ongoing technological developments, will exert a long-term downward pressure on prices[3], see the suggested development of the index. Only a continuing series of major disruptions could halt that trend. Introducing new taxes on energy could halt price erosion for users and would, comparatively speaking, lower energy use and the emissions related to it. Both the European Community and the Dutch government are contemplating the introduction of a general energy tax to increase the price of energy again and thus decrease its use and the emissions related to it. This, at first glance, seems a sensible option for environmental

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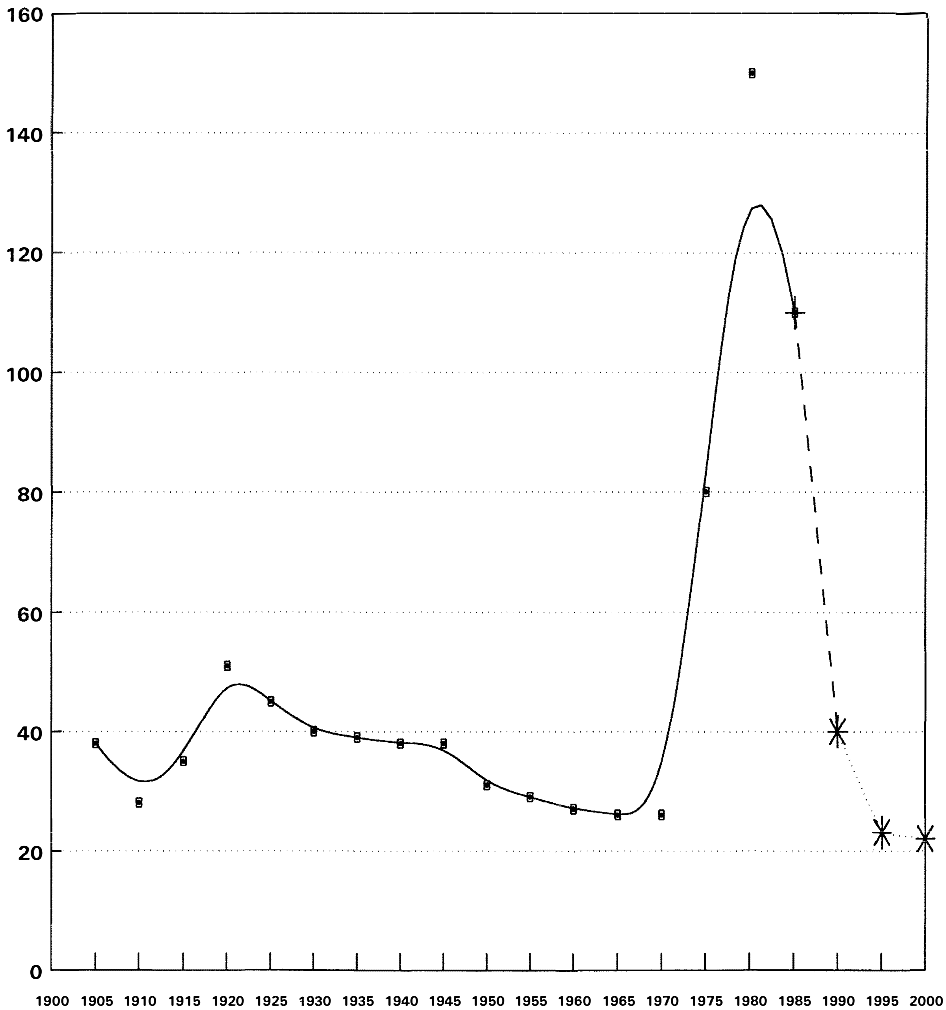
[1] This chapter draws heavily on a paper prepared for a workshop organized by the Wiardi Beckman Foundation on Dutch proposals for energy taxes investigated by the "Wolfson Commission", see Huppes 1992.

[2] Gibbons et al. (1989) give similar figures of energy per unit of GDP. Manne and Richels (1991) give a smaller difference between the US and the former Soviet Union. Both the GDP measures and the energy definitions may differ between studies. "Europe" is covered in them under "other OECD", including Canada and Australia. Their figures indicate that China uses at least eight times more energy per unit of GDP than the OECD without the US.

[3] CPB 1992 and Wolfson Commission 1992 assume an autonomous rise in prices for the next decades of around three percent per annum. This assumption is based on a supply-demand model which has only a very limited ability to take into account technological development. Costs of oil and gas exploration and exploitation have decreased dramatically in the last decade. Main sources of decreasing costs are better modelling of results of seismic tests and horizontal drilling, according to several Shell publications and to Odell (personal communication).

policy. Long-term practice has established both its administrative applicability and its environmental effectiveness. At the same time, the position developed in Part Three is that such a product tax is at best a second- or third-choice option. Could the first-choice options of the flexible response strategy be more attractive in terms of environmental effectiveness, costs and equality? Could implementation be as easy? To that question this chapter will attempt to give an answer.

FIGURE 5.5.1 OIL PRICES IN THE TWENTIETH CENTURY[1]



[1] Source: Radetzki 1990. Data for 1990 own estimate, for 1995 and 2000 linear extrapolation excluding the peak data of the Seventies and Eighties.

In this fourth case on policy design I will first direct attention to the practical proposals on energy taxes and on their potential for environmental policy, in section 5.5.2. See Appendix 5.5.1 to this chapter for comments on the most detailed analysis of the introduction of energy taxes available, as produced in a study by the Wolfson Commission. Section 5.5.3 treats two main problems related to energy use in more detail; energy depletion and global warming. Appendix 5.5.2 gives an estimate on the availability of energy. It first indicates that any signs of depletion of fossil and fissile energy sources is extremely unlikely for many centuries to come. Long before that depletion might occur, the price of solar energy will make these resources irrelevant for most energy production. There thus is no energy depletion problem connected to the use of fossil energy sources but a global warming problem and a number of other problems. The following sections then focus on the global warming effects of carbon compounds in the structure familiar from the preceding two case studies on substances:

◇ substance flow analysis, especially of the economic flows of the relevant compounds (5.5.4)
◇ technical options for emission reduction (5.5.5)
◇ policy instruments applicable (5.5.6)
◇ applicability of the flexible response strategy (5.5.7).

The chapter ends with conclusions on the case (5.5.8).

5.5.2 Energy taxes as a main subject of environmental policy

In the discussions on ecotaxes that started in Germany in the late Seventies energy taxes play an important role, see Teufel et al. (1989) and von Weizsäcker (1992). The central aim of these ecotaxes is to shift the tax burden from taxes on labour to taxes on activities that harm the environment. More recently, the LMO, an influential Dutch environmental organisation, submitted detailed proposals for such ecotaxes (Nentjes et al. 1990; LMO 1991). According to their computation with a sectoral model the energy tax would, after behavioral adjustments, account for more than fifty percent of the proceeds of all ecotaxes proposed. With the extra taxes on road traffic the total share of energy taxes in total proceeds is over eighty-five percent (LMO 1991 p.22). Most of the other taxes proposed relate to single environmental problems, such as those of pesticides, nutrients, volatile organic substances and ground water. In their view the energy tax especially relates to CO_2 emissions. Its form, however, is a product tax, for ease of administrative application and because it also relates to many other environmental problems.

The general idea seems that if energy-demanding activities diminish, many problems together will diminish as well. Energy saving then is a catch-all with extreme simplicity of administration and an extremely broad range of beneficial environmental consequences[1]. Energy use, so the reasoning goes, is directly related to a number of environmental interferences and similar problems:

[1] In a study for the Central Advisory Board on the Environment we distinguished four scenarios. Two lead to disaster, the *Titanic scenario* because nothing effective was done and the *Coercion scenario* that shows a collapse of legitimacy and an ultimate breakdown of political decision-making. The two scenarios that might work are the *Flexible response scenario*, which is specified in more detail in this study, and the *Spearhead scenario* which concentrates on volume restrictions in the environmentally worst economic sectors, including the energy sector. See van Manen et al. 1990.

◇ It is a central element in resource depletion
◇ It is the main source of pollution with CO_2, SO_x and NO_x, and, to a lesser extent, of heavy metals
◇ Transport, an extensive energy-demanding activity, has a ravaging effect on landscape and ecosystems.

Thus there are apparently reasons enough to single out energy in environmental policy, with taxation as a main instrument. Energy savings that become attractive because of the induced higher energy price, would save on many other environmental effects as well.

This seemingly attractive picture has a number of flaws. First, energy depletion itself is not a problem at all, as will be indicated in Appendix 5.5.2.

Secondly, when taxing energy instead of the pollution that is the problem, no incentive for pollution prevention is created other than that related to reducing energy use. Technologies for the reduction of acidifying emissions of carbon, sulfur and nitrogen are not stimulated. The mechanism leading to emission reductions and other effects is the most expensive available: volume reduction. Substantial reductions in volume proportionally diminish the value created by the now reduced activities. Experience has shown that with most emission problems techniques are available to substantially reduce emissions at costs that are a small fraction of the value of the activity that caused the former emissions. In electricity production from coal one of the most abundant resources, emissions of SO_x and NO_x together may be reduced to virtually zero in a combined cycle coal gasifier. These coal gasifiers are very nearly commercially attractive. Specific emission taxes on these substances would easily create an advantage for that zero-emission process. Thus, the analytic approach of acting directly on the emissions caused would be much more attractive in terms of costs and efficiency. It would have a limited effect on energy volumes produced and consumed. It would have a much higher effect on the environmental problems that are to be solved than ever could be created through volume reductions in energy consumption. The example of the coal gasifier may further illuminate this position.

For CO_2 emissions, techniques of emission reduction are currently being investigated. One method is to store the CO_2 underground, in depleted gas reservoirs. This technique is most easily added to a combined cycle coal gasifier using pure oxygen for burning. This gasifier produces CO_2 in a very concentrated form. In such an installation all emissions resulting from the combustion of coal, one of the most contaminated of all energy resources, fall to nearly zero. The total energy efficiency is a bit lower than that of current coal furnaces that reduce only eighty percent of SO_x and NO_x emissions and do not reduce the emission of CO_2 at all[1]. The costs per kWh are a bit higher. See Lucht 91 (1990) for exact figures and section 5.5.6 for more data. The central point of the

[1] The addition of CO_2 removal and disposal would lower the overall efficiency of electricity production in the combined cycle installation being built in the Netherlands from 43% to 33%, see Lucht 91. The reduction of process emissions of CO_2, NO_x and SO_x is bought by a higher direct resource use of coal of nearly 25% per kWh. Indirectly, taking into account other processes in the life cycle, the effects might be less, or even more extreme. Extra coal production required for CO_2 removal also releases more CH_4, a much more potent climate gas than CO_2. The extra installations also require more steel which is produced at substantial emissions of NO_x and SO_x. If this line of reasoning is pursued the character of the analysis changes from the one-problem oriented substance approach into the life cycle of product approach, the product systems compared being different alternatives for producing, as a functional unit, a certain amount of electricity.

argument here is that, whatever the exact figures may be, *no level of the energy tax will ever be able to make the zero-emission alternative of coal-based electricity production commercially attractive*. The price increase in energy would only reinforce the price differential, due to the slightly lower net efficiency of the combined cycle process with CO_2 storage. Comparatively modest taxes on emissions would easily create the price differential that makes the cleaner process profitable.

Thirdly, it is not true that energy saving leads to almost proportional reductions in all effects. Increasing the fuel efficiency of cars for example, will not diminish the disturbances caused by traffic.
Fourthly, if volume reductions in one type of activity occur while the amount of labour and its productivity remain the same, other activities will grow compensatingly. It is not at all sure that the increased other activities involving the same amount of spending, are more beneficial on average environmentally seen.
Sixth, why should it be oil that is taxed this way? Might not other options be more attractive, like taxing iron and aluminium, or concrete? Diminishing the activities in which these materials are involved would seen at least as attractive. The first five arguments given are hold against these latter taxes as well.
The conclusion here is that it is by no means self evident that taxing energy, as an input tax on a broadly used resource is beneficial, let alone that it is an ideal means for general environmental policy.

Current proposals developed by the EC and the Dutch government, have a dual basis. One part of the energy tax relates to the energy content of the resource, the other part relates to the carbon content. These proposals thus seem to concentrate on two environmental problems, energy depletion and global warming. Other proposals seem similarly mixed, at least in intention. Becht (1989) advocates a variable tax that increases the price of fuels to the level where flow energy becomes commercially attractive, stipulating climate problems as the main reason. The basis for taxing is then not only the energy content but also the costs of producing secondary energy with non-renewable energy resources and the costs of renewable energy. Manne and Richels (1991) similarly compute a switch over price based on the switch to non-fossil energy. See Kverndokk (1992) for the resulting figures for the OECD and the (former) Communist countries. The difference with Becht, i.e. the difference between 'non-renewable' and 'fossil', is that Becht also taxes nuclear energy out of the market. These switch over prices may differ due to environmental requirements and other factors mentioned, but not in relation to the amount of carbon per MJ of resource. Teufel et al. (1989) and Flaving and Dunning (1988), on the other hand, advocate taxes based on carbon content only. So does the Congressional Budget Office (1990) which aims solely at the emission reduction of CO_2[1].

All specific proposals adopt the problem oriented approach, not the catch-all concept that singles out volume reductions in one sector, the energy sector. All proposals presuppose that other policies take care of other emissions leading to other problems. The proposals only indicate or specify the effects on the depletion of energy resources and on CO_2 emissions. It may be noted that the sectoral catch-all approach has some similarity to the

[1] See Appendix 5.1 for the levels of the taxes advocated.

life cycle analysis approach, that also considers all environmental effects together. The difference is that sectoral reductions imply a reduction in volumes consumed, while at the product level a changeover to functionally similar products or installations is the alternative. Such changes generally need not imply high costs, as do volume reductions. In the next section I will examine the two main problems involved in the energy tax proposal in more detail, before proceeding to the substance flow analysis and to possible solutions through the flexible response strategy.

5.5.3 Energy depletion and global warming: the problems

How serious are the problems of energy depletion (1) and global warming (2) and how exactly are they related to energy use? This section is dedicated to these two questions, and also builds on the results of a rough investigation on the availability of energy sources in Appendix 5.5.2 at the end of this chapter.

1. Energy depletion

To assess the seriousness of the depletion of fossil and fissile energy resources, first, their recoverable stocks have to be quantified in relation to some level of use. Not only these stocks are relevant. All other sources of usable energy are relevant for this analysis, especially those based on permanent flows of energy such as solar power and heat from the earth core. When does a resource restriction become so serious as to form an environmental problem? If one really goes to the extreme, any non-renewable stocks are by definition finite. Their use thus contributes to depletion as a problem for some future generation. However, if alternatives are in principle available and if stocks will not be depleted for several thousand years, the solution to such a problem would have no priority now.

How large are the stocks of depletable energy resources, absolutely and in term of years of a certain supply? How important can the alternative sources of flow energy be, quantitatively? Suppose that for an extremely long period of ten thousand years the current level of energy use could be supplied, i.e. a yearly use of about 300EJ (exajoule). It is shown in Appendix 5.5.2 that this is possible in several ways which are not mutually exclusive. It would be possible with fossil energy alone. It would be possible with fissile resources, using breeder technology. In that case at least three times current energy use could be supplied for 10.000 years. The uncertain possibilities for the development of fusion power have not been taken into account. Current demand could also be met for an indefinite period by flow energy, with a predominant role for direct solar power. Combined, a long-term energy use of over ten times the current amount is possible for many thousands of years, see table 5.5.1. In estimating ultimate resources one should be aware of confusing proven stocks of any magnitude related to ultimate stocks. Proven stocks tend to remain constant in time in number of years of current use available. They are the "work-in-progress" amount of resource extraction. See in this sense Simon (1988) and for some more historical data Deadman and Turner (1988, pp.70-1).

TABLE 5.5.1 ESTIMATES OF ENERGY RESERVES AND POTENTIAL LONG-TERM ENERGY PRODUCTION, IN EJ ($=10^{18}$J) THERMAL, GROSS SOURCE: APPENDIX 5.5.2

ENERGY SOURCE	LONG-TERM ESTIMATE OF RESERVES	POTENTIAL PRODUCTION PER YEAR OVER 10,000 YEARS
fossil energy[1]	3,500,000	350
nuclear energy[2]	10,050,000	1,005
flow energy	indefinite	1,850
sum total	13,550,000	**3,350**

Hotelling has formulated rules for an optimal path towards depletion. These rules, widely accepted among environmental and resource economists, indicate that a price level should be realised that keeps the value of the resource amount not yet extracted equal, for example in monetary terms. The amount ultimately available must therefore be known. Applying this Hotelling rule to the energy situation depicted here has clear results. Suppose that all non-fossil energy production options are disregarded[3]. Current use of fossil energy is only in the order of 0.01% of the ultimate stock per year. Thus, price increases of 0.01% per year are due for depletion reasons, on top of those required to cover higher costs of extraction. It is clear that such price increases are quite irrelevant to any practical choices now.

Unfortunately this is no reason for joy and jubilation. The fossil reserves are there. If they are used in the current manner, without real limitations on emissions, the world will become a less hospitable place. Because of emissions of CO_2 and methane, the Netherlands will become a tropical country, at least those parts still above sea level. All other emissions related to fossil energy consumption, such as oxides of nitrogen and sulfur, and several trace elements, may make life for man, animal and plant unpleasant, unhealthy or even impossible. It does not seem wise to use the limited capacities for regulation on a problem so improbable as energy depletion, while so many risks on real calamities and hazards still remain, such as those of substantial climate change.

2. Global warming

There is a near universal consensus that global warming, if it does occur, is a serious problem. If substantial climate change occurs, the consequences will be catastrophic in many places. Many lives will be lost, nature areas and species will disappear and the costs to adapt to the changed environment will be very high. The main mechanisms responsible for the global warming effect are undisputed. Molecules trap radiation selectively. The inflowing radiation from the sun reaches the lower atmosphere and the surface of the earth. If it is not radiated back into space the earth will slowly heat, an effect also due to energy released by the earth itself from radioactive decay. The incoming radiation from the sun has a shorter wavelength on average than the radiation

1 Excluding gaseous chlatrates, the single largest stock on earth, including shale oils.
2 Excluding fusion power.
3 Hotelling's rule does not directly apply to the additional possibility of flow energy.

flowing from the earth into space. The latter is mainly in the infrared range. Molecules that trap radiation in the infrared range, but not in the shorter wavelengths, prevent radiation into space to a certain extent. If their concentration in the atmosphere rises, more energy will flow in than flows out and the global warming effect starts. It will go on until a new equilibrium temperature if any is reached. The trapping capacities of each molecule for all wavelengths are unquestionably known.

What happens to a molecule emitted is still an open question, however. CO_2, a major greenhouse gas, is an example. In the extreme long term any extra CO_2 emitted will be primarily stored at the bottom of the sea. Shellfish and fish will bind the CO_2 as calcium carbonate that sinks to the bottoms of the sea at the end of their lives. No exact predictions are available of what makes up the extreme long term. Several hundreds to several thousand years is the range. In the less distant future but still in the long term, most CO_2 will be absorbed into the seas in the long carbon cycle, both by biotic and a-biotic processes. Effects on climate become very much dampened within this time framework. The actual duration of this long term is open to much debate, but it will be in the order of centuries. The speed of the dampening process decides how much CO_2 may be emitted without leading to unacceptable climate change. The main debate, however, is not on this second long-term set of effects either. It is on the effects in the time range from the next decades to well into the next century, this being a short period in terms of climate change.

In the still shorter term there are many relations that may reinforce, dampen or even reverse effects of a climate gas on climate, as two extreme options and one in between. The one extreme is the reinforcement of the global warming effect on climate by secondary effects. After a point of no return, reinforcing tendencies take over. A moderate temperature rise thaws the permafrost, releasing huge amounts of CH_4, a climate gas that per molecule is twenty times as potent as CO_2 in trapping heat measured over a century but a thousand times as potent when measured over a year. Wetland at sea level, now containing a huge amount of biomass, is swallowed by the sea, giving off the main part of its carbon content. Their high biological carbon fixing potential is lost. It is replaced by the biological carbon fixing potential of the sea that, on a square meter basis, is comparable to that of a tropical desert. Next, the increased temperature leads to an increased evaporation from the sea. Water being the most potent climate gas, more heat becomes trapped, with an ever increasing global temperature as a consequence.

Dampening mechanisms might prevent the occurrence of either extreme. The adsorption by the sea of carbon from the atmosphere is the main dampening mechanism of the global climate system. It also is a major subject of debate. An international programme to measure flows of carbon in the upper layer of the oceans was started only in 1989[1]. Generalisations from only locally available measurements are extremely difficult. Moreover, the increased growth of plants due to higher concentrations of CO_2 will fix more carbon, especially in roots. The increased albedo of the earth due to cloud forming is a third dampening mechanism.

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[1] The Joint Global Ocean Flux Studies (JGOFS) combines satellite measurements of large areas with local measurements from ships. The main aim is to better understand and model the role of biological processes in carbon uptake and transport between different layers of the seas. See Gillis (1992).

The other extreme is a reversed effect on climate. Initial temperature increases may lead to desertification in the tropics, especially after the forests there are levelled. This may lead to a structural increase in the albedo of the earth. Incoming radiation is much more reflected and reflected much more in the unconstrained shorter wavelengths than in the infrared. A prolonged cooling period would become inevitable.

If either of the extremes becomes dominant, these chaotic short-term effects may dominate the long term. Such an irreversibility is possible, even if not yet probable. Other types of effects may counteract this potential instability. A pulse of carbon emissions would accompany the decay of that biomass, setting in motion the mechanisms that lead to ever increasing global temperatures. Models being limited and empirical data being limited, different models will predict different effects for the same emission levels of climate gases. However, even if major mechanisms remain disputed there is good reason for active long-term management. Passing an unknown threshold might lead to runaway effects on climate. A not entirely improbable scenario has been developed by Greenpeace in which things go wrong irreversibly, especially if ecological effects of ozone layer depletion are also taken into account. See Leggett (1992). The small but positive chance of such a catastrophe, with stable global warming more probable than unstable global cooling, is reason enough for action now.

Indirect reasoning might be more important in assessing the probability of instability than the imperfect models now available. If the climate system were really unstable it would long ago have left the range it is in, for example due to one of the major catastrophes that hits the earth regularly. Large eruptions of volcanoes have occurred regularly in the geological history of the earth. Such eruptions may cause a cooling period by shielding the radiation of the sun through sulfur, carbon and dust emitted to the upper parts of the atmosphere. A large portion of all living biomass dies,[1] leading to desertification and consequently to a higher albedo of the earth. Still, the earth recovered to current temperatures. The system thus, to the cooling side, cannot be totally unstable in the long run. If a return to current temperatures would occur soon may be doubted however. The current state of climate affairs is quite exceptional, being named an interglacial for that reason. The stability at the warming side cannot so easily be inferred from the historical record. The amount of CO_2 in the atmosphere is surpassing known historical values and thus leaves the range of the predicting power of the past. On that side the current imperfect models may give the best answers available.

Even if global effects would match between models, the regional effects may be assessed very differently. NASA's Godard Institute of Space Studies has developed a climate model. A global increase in temperature of several degrees Celsius would have a net effect on US grain production of zero. The model of the British Meteorological Office, on the other hand, predicts drought and declining harvests, especially in the US, see Pearce (1992b).

All the same we are living in a climate similar to that of the last ten millennia, neither stuck in the cold, nor heated to flooding. Of course, this is no proof that the emission of

[1] Such a global winter may also be due to a large nuclear war or the impact of a large celestial body.

climate gases does not lead to catastrophes. Current CO_2 levels in the atmosphere are substantially higher than the peaks associated with warm interglacials in the last hundred thousand years. The levels of such man-made global warming gases as CFCs are still rising. The relative stability of the past only gives reason for hope that some built-in stabilizers are to some extent effective. If structural long-term policy keeps down the emissions of climate gases, effects on climate may remain limited. If the real short term may be overcome by chance - any action now is quite irrelevant for what happens to climate in the next years - then the longer term has to be analyzed and managed.

What are the conclusions here for the quantified problem analysis and the development of technical and policy measures? First, it remains undisputed that further emission of greenhouse gases increases risks of unstable climate reactions. Even if only temporary, they still may cause immeasurable human suffering and death, a substantial decrease in biodiversity, and high costs. Short-term measures are thus appropriate. These could include diverse activities. Bringing iron into the southern oceans, for example, would create a pulse of algal growth that would balance the carbon budget for some time (proposed by James Martin, as cited in Gillis 1992). Bringing soot and sulfur in the upper layers of the atmosphere would reduce the incoming radiation from the sun. Other not purely technical measures would be socially extremely difficult to implement, such as stopping logging for some years and stopping all deforestation now. Such short-term measures, those of a technical nature often being disputable because of other risks created, are not the subject of policy development in this chapter.

Secondly, given current levels of emissions, a long-term shift in climate is highly probable, although there is much debate about quantitative relations and timing. Even a stable shift in climate has extreme consequences for a substantial portion of the human population. Some regions will be flooded by the sea, some humid regions will become arid, and weather conditions will become more unstable.

Thirdly, it is all greenhouse gases together that cause the climate problem, not a single one. It does not make sense to single out one substance for policy development. The emphasis on only carbon, in only energy resources, because of emissions of CO_2, is unbalanced, especially in the short term. CO_2 from energy resources now accounts for less than forty percent of the human contribution to global warming, according to the IPCC, see table 5.5.2. Problem shifting may easily occur. Shifts from coal to natural gas, induced by carbon taxes on energy, indeed reduce CO_2 emissions. However, in many countries this reduction of the climate problem may easily be offset by extra methane leaking from gas pipes before burning, see Lelieveld et al. (1991)[1].

Fourthly, policy development cannot be based on established predictions. Only uncertain chances of outcomes form the basis. Quantification of effects in economic terms is impossible. A choice has to be made of an acceptable level of risks and an acceptable level of probable climatic change.

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[1] Methane leaked from coal mining may counteract this effect. There are substantial differences between coal mines in emissions of CH_4.

TABLE 5.5.2 RELATIVE CONTRIBUTIONS OF DIFFERENT GASES TO GLOBAL WARMING, BASED ON GWP_{100}, GROSS EMISSIONS 1990. SOURCE: HOUGHTON ET AL. 1990, P.61

Trace gas	Emissions (full molecule weight, 10^9 kg)	Proportion of GWP_{100}
CO_2	26000	61
CO	200	1
CH_4	300	15
N_2O	6	4
NO_x	66	6
CFCs	mixed	11
Others	mixed	1

Fifthly, if the short term is overcome, deliberately or by sheer luck, it is the structural, long-term management of economic inflow and outflow that has to be taken care of. The analysis here concentrates on this longer time horizon. The choice of substances included in the analysis should be adapted to this time horizon.

Long-term analysis

Which climate gases are relevant for the long-term analysis? Let us assume that CFCs and halones and a large proportion of HCFCs will be phased out by the beginning of the next century. This will happen because of their effects on the ozone layer, not for climatic reasons. The HCFCs not yet phased out partly take their place. If these will contribute substantially to the global warming problem, their contribution to ozone depletion will be substantial as well. It may also be assumed here for practical reasons that future emissions of these substances will not contribute much to the climate problem[1]. Main contributions to the climate problem thus originate directly from four compounds, CO_2, CH_4, NO_x and N_2O. The latter two compounds do not enter the analysis here for purely practical reasons[2]. As already pointed out the former two substances should be analyzed together.

However, the substance flow analysis will not be limited to CO_2 and CH_4, although it is only these compounds of carbon that contribute to climate change. Any form of carbon compound that may readily be formed from these two gases *and* formed back into them will be included in the analysis. For the long-term time scale this of course includes all organic materials containing carbon. Included also are unstable inorganic compounds, especially the carbonate ions that may be formed from soluble sodium and potassium (hydro) carbonates. Stable forms such as calcium carbonate are not included in the flow analysis. Only if these decompose, their metabolites, mainly CO_2, enter the carbon cycle thus defined for the long term[3]. That broader analysis includes carbon-in-energy. Other sources and sinks, here defined as structural inflows and outflows, are also taken into

[1] Past emissions will stay in the atmosphere for centuries.

[2] One practical problem with N_2O is that its sources are not well quantified, see Houghton (1990).

[3] Odum (1989, pp.123-4), for example, includes calcium carbonate in the analysis of the cycle.

account. A full analysis of the economic flows of all substances that form part of the carbon cycle is given in section 5.5.4.

Indirectly, still other types of emissions may contribute to the greenhouse problem as well. Acid emissions break down carbonates, releasing CO_2 in the process. GWP is also based on secondary effects in the environment. SO_2 would thus directly have a negative global warming potential by itself and a positive indirect potential because of the extra weathering of carbonates. No data are available. This quite reasonable line of reasoning will not be followed any further here.

Conclusions on global warming

Global warming, as a major environmental problem, may quite legitimately be an independent focus of attention in policy design. In developing options for solving that problem, it seems illogical to put so much emphasis on carbon and energy. Each and every contribution to reducing global warming emissions should be considered comparably, not only for environmental reasons, but also for reasons of justice, effectiveness and costs. In partial analysis and partial solutions, problem-shifting could seriously play havoc with all these criteria. In the long-term analysis that follows, all substances contributing to the carbon cycle are considered. N_2O is omitted from the analysis here for practical reasons only but should be included, as should other emissions affecting the atmospheric temperature such as (H)CFCs and sulfur dioxide. No technical measures and policy instruments will be developed for such substances.

5.5.4 Substance flow analysis of climate changing carbon: CO_2 and CH_4

The analysis of climate changing emissions and the flows of the related substances into and through society, is restricted here to carbon compounds, assuming that other climate changing emissions are treated similarly (mainly nitrous oxide and HCFCs) or are phased out by other means (CFCs and halons). If not, shifts to other climate changing emissions might occur[1]. The analysis is made at the global level, although there is no global administration. For purely global problems this seems a good start. Moreover, if national policies are pursued, as in the previous examples of energy taxes, it is the nett effect at the global level that has to be taken into account. This would require a global analysis anyway. The substance flow analysis at the global level is also less complicated. The following analysis concentrates on the flows in and through the economy of the world. The flows through the environment, and also the resulting concentrations in the atmosphere, are taken from the literature in an aggregated form.

The carbon cycle may be analyzed on different time scales. The short cycle is biological. In it biological processes are dominant. Plants transform CO_2 from the atmosphere and from lakes and seas into carbohydrates. All creatures feeding on plants, from single cell moulds and bacteria to higher animals, transform the carbohydrates and derived compounds back into CO_2. The process is not fully efficient, however; the cycle is not closed. Some waste is deposited. The main deposit is at the bottom of the seas, such as calcium carbonates from shells and bones. The other deposit is in the form of organic matter that is not broken down, both in the seas and on land. These flows of carbon are

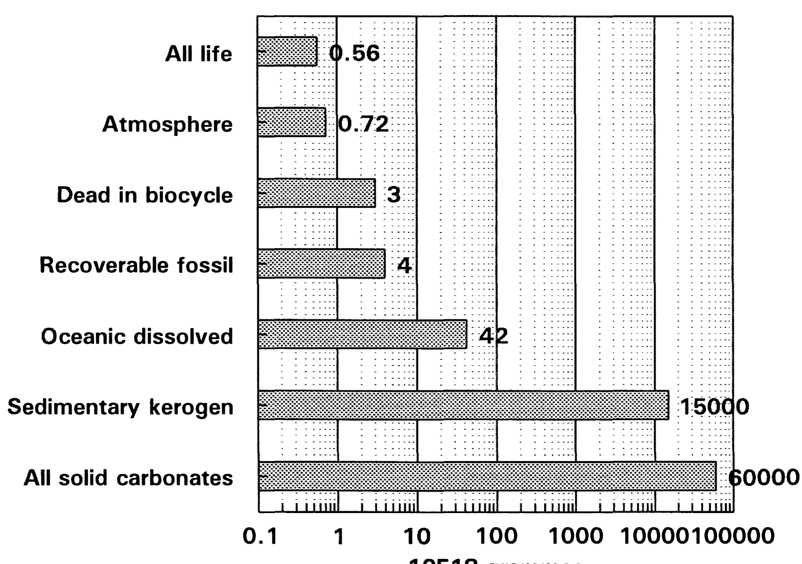
[1] Shifts to other problems will occur as well, if no policy is developed for these problems.

not recycled biologically. Some older deposits of carbon are activated again, more or less supplementing the current losses.

On the extremely long geo-chemical time scale, covering up to hundreds of millions of years, the carbon cycle is mainly closed. Geological processes close the cycle, quite independently from any human activity like fossil energy use. Two types of processes play a central role in it, both related to the tectonics of the earth. First, the carbonates deposited subduct under the continents as the sea floor spreads from the mid-ocean rigs. The carbonates are heated under high pressure, primarily forming silicates and CO_2. The CO_2 surfaces through vents and volcanic eruptions, closing part of the cycle. Secondly, the carbonates and the remains of organic matter not deposited under the continents may surface, by tectonic movements again, and by erosion and weathering of top layers. The carbon contained in them will then take part in the biological carbon cycle again, through weathering and oxidation.

The differences between the amounts of carbon involved in both cycles are tremendous. All living organisms together contain less than one tenth, of a percent, of a percent (10^{-5}), of the carbon circulating in the long cycle. Total atmospheric carbon is there in similar amounts. Dead organic material circulating in the biological cycle and recoverable fossil fuels are each almost an order of magnitude more greater. Dissolved carbon in the oceans is nearly two orders of magnitude larger than carbon in all living organisms. It still is not more than half a percent, of a percent, of the total amount in the long cycle. See figure 5.5.2, with data from Berner et al. (1989).

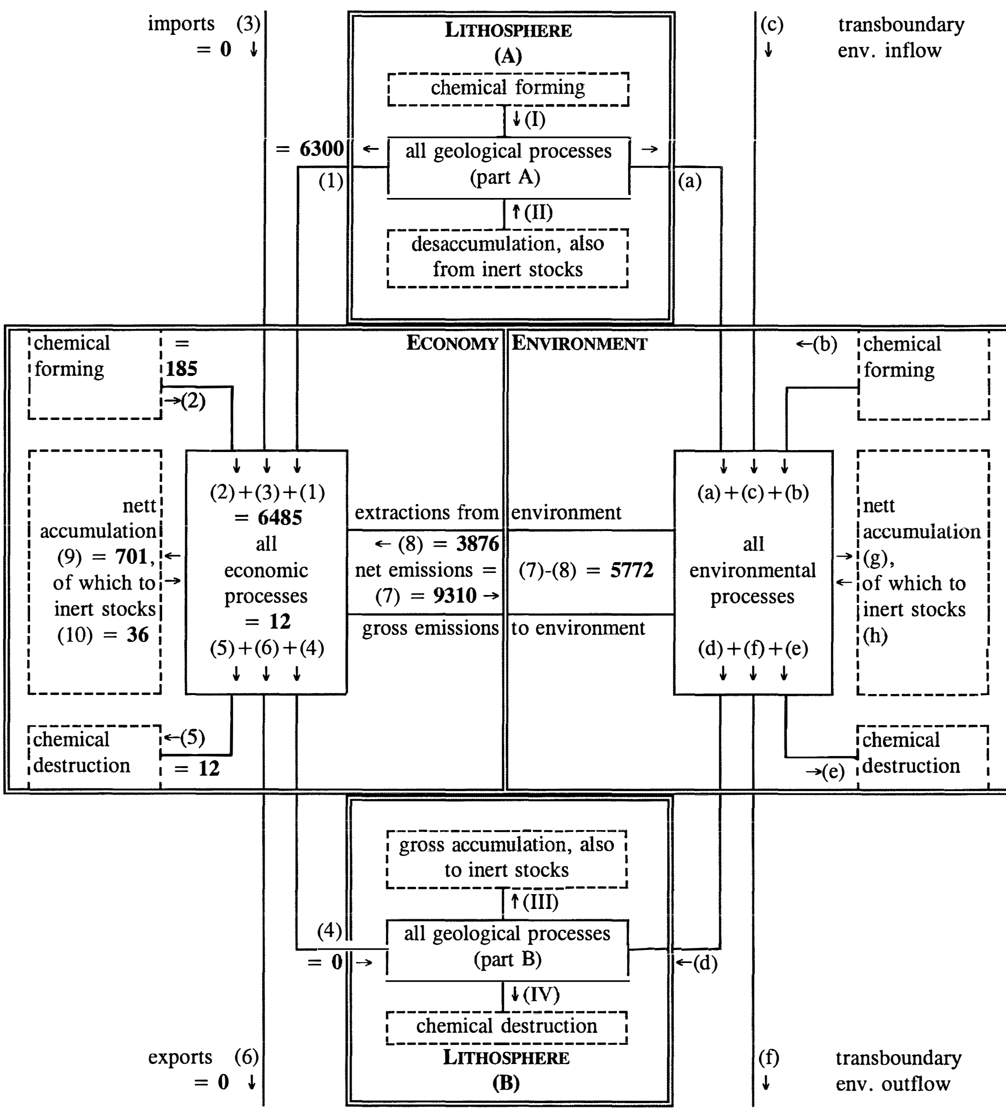
FIGURE 5.5.2 AMOUNTS OF CARBON IN THE GEOCHEMICAL CARBON CYCLE, LOGARITHMIC SCALE[1]



[1] Source: Berner and Lasaga, 1989.

The analysis presented here aims at bringing in sight all human contributions to the long-term global carbon cycle. These human contributions are *not* on a geological time scale, hence not all mechanisms are relevant for the analysis. The transformations within the crust of the earth, especially the transformation of calcium silicates into calcium-magnesium carbonates and the transformation of carbonates into silicates, remain undisturbed by current human activities. Measures that only influence the biological cycle are disregarded as well, however important these might be to the short-term development

FIGURE 5.5.3 GLOBAL INFLOWS, OUTFLOWS, NETT EMISSIONS, AND ACCUMULATIONS OF CARBON, IN 10^9 KG C, DATA FOR FLOWS IN THE ECONOMY ONLY



of climate. Concentrating on these structural human contributions gives a criterion for which carbon compounds to include in the analysis. The analysis of yearly flows singles out those chemical forms of carbon that figure in the global non-human carbon cycle. It is not just CO_2, but all chemical forms of carbon that are readily transformed into CO_2 in the time scale chosen. These include all organic compounds. Wastes from food production are usually oxidized into CO_2 within decades. Moreover, all inorganic compounds are included that usually will transform into either CO_2 or organic compounds. Carbon in steel is an example. In the long run, most of it will rust, freeing the carbon. Some parts of the flow analysis are a bit odd. The human body is part of the economy. If it is viewed as part of the environment, eating would be an emission, while the sewer would extract substances from the environment. Thus internal human food processing is an economic process here.

TABLE 5.5.3 GLOBAL INFLOWS, OUTFLOWS, NETT EMISSIONS, AND ACCUMULATIONS OF CARBON, IN MILLIONS OF TONNES (10^9 KG), DETAILED DATA FOR ECONOMIC FLOWS ONLY. SOURCE: APPENDIX 5.5.3

1[1]. Extraction from lithosphere	abs.	perc[2].
-Extraction of:		
*oil	2432	37.5%
*ultra heavy oils, tar, shales		p.m.
*natural gas (methane)	1209	18.6%
*all coal	2362	36.4%
*peat	2	0.0%
-Preproduction losses of:		
*methane from *methane	45	0.7%
*coal	35	0.5%
*oil	14	0.2%
*other fossil	24	0.4%
*oil	24	0.4%
*coal	23	0.3%
-Erosion	130	2.0%
Subtotal from lithosphere	*6300*	*97.1%*
Subtotal to lithosphere		
Net inflow from lithosphere	**6300**	**97.1%**

4. Deliverance to lithosphere	
-Disposal in lithosphere	
*CO_2 in natural gas domes	0
*disposal of CO_2 in deep seas	0
*?	
Subtotal to lithosphere	*0*

2. Chemical forming in the economy		
-Glowing calcium carbonates		
*cement production	123	1.9%
*plaster production	24	0.3%
*CaO in diverse industrial appl.	1	0.0%
-Binding sulfur with lime		
*all processes	4	0.0%
-Methane from organic wastes	33	0.5%
Subtotal chemical forming	*185*	*2.8%*
Subtotal chemical destruction	12	0.1%
Net chemical forming	**173**	**2.7%**

5. Chemical destruction	
-Forming of carbonates	
*weathering of concrete	12
-Other stable inorganic compounds	
*carborum	?
*?	
-Stable organic compounds	
Subtotal chemical destruction	*12*

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1 Numbers correspond to those in the flow scheme.

2 As a percentage of total structural inflow, as extraction from lithosphere plus chemical forming. There are no imports and exports at the global level.

Total structural inflow	*6485*	*100.0%*
Total structural outflow	12	0.1%
Net structural inflow	**6473**	**99.8%**

Total structural outflow	*12*

8. Extraction from the environment[1]	
-Biotic flows, fixing CO_2	
*wood fuel	682
pulp	82
construction and other	557
*fish and game	11
*grains	174
*grass	505
*other food and fodder	78
*rice roots and stabs in paddy fields	45
-Biotic flows, taking organically bound carbon from the environment	
*deforestation	1600
*erosion of top soils	130
-A-biotic flows	
*CO_2 fixation in concrete	12
* ?	
Subtotal extractions from environment	*3876*

7. Emissions to the environment		related to: CO_2	CH_4
-Combustion processes			
*fossil fuels		5947	?
*wood		662	20
*wastes	plastics	1	?
	organic	17	?
-Biotic processes			
*from ruminants directly			60
*rice growing in paddy fields			45
*organic materials (landfill)			45
-Chemical forming			
*cement		123	
*plaster		24	
*industrial. applic.		1	
*oxidation of carbon in steel		0	
-Organic materials			
*biomass, faeces+hum.resp.		69	?
	manure+anim.resp.	609	?
*pre-production losses:	oil	24	
	coal	23	
*fossil methane spills			118
-Net oxidation of agric. soils			
*erosion		260	?
-Deforestation		1550	50
Gross emissions to environm.	*CO_2*	*9310*	
	CH_4		*338*
	Total carbon		*9648*
Extractions from environm.	CO_2	3876	
	CH_4		0
	Total carbon		3876
Net emissions to environm.	**CO_2**	**5434**	
	CH_4		**338**
	Total carbon		**5772**

Total inflow into the economy:	
*from lithosphere	6300
*chemical forming	185
*imports	none
*from environment	3876
Total inflow	*10361*
Total outflow	9660
Accumulation on balance	**701**

Total outflow out of the economy:	
*to lithosphere	0
*chemical destruction	12
*exports	none
*to environment	9648
Total outflow	*9660*

[1] This inflow is CO_2 related only, no methane is involved.

9. Accumulation in the economy		*from balance:701*
In products and installations		
plastics		45
wood		557
paper		43
carbon in steel		7
other stable carbon compounds		0
Inert stocks (10)		
land fill	biotic organic	30
	plastic	5
	C in steel	1
Subtotal accumulation		*688*
Rounding off errors, mistakes, inconsistent data, = balancing item		13
Total accumulation		**701**

Some large flows of carbon are excluded from the analysis. Marl, for example, consists mainly of calcium carbonate ($CaCO_3$). The very large amounts extracted, e.g. as road building material, are excluded by this criterion. Only in the extremely long geological time scale will calcium carbonate, first deposited, be weathered away into CO_2 again. However, if marl is externally heated to produce calcium oxide (CaO), the carbon part comes free as CO_2. This forms an inflow of carbon into the economy. In the production of cement, calcium plaster and steel this CO_2 is emitted directly into the atmosphere. However, the carbon from marl may also be bound with sodium in certain production processes for soda ($Na_2CO_3.10H_2O$). The insoluble, stable calcium carbonate, itself not in the flow analysis, is transformed there in the soluble and highly reactive form of soda. The latter substance would thus be included in the flow analysis[1]. Excluding the very large amount of carbon in marl from the analysis cleans the data from 'noise' that itself is irrelevant to the greenhouse problem. Marl dug for road building is completely excluded from the analysis; it is not even an inert stock. This systems level of analysis does create some demarcation problems. In cement production CO_2 is emitted in quite large amounts. The carbon atoms emitted have already been registered as an inflow for those carbon atoms originating from fossil fuels. The heating of marl also contributes to CO_2 emissions by freeing the carbonate part from its bonds to calcium. Thus, at current cement and steel production, there is an inflow of carbon from marl into the economy and a directly corresponding outflow of (part of) that carbon from the economy to the environment, in addition to the inflow of carbon in fuels and their emission at burning. That distinction cannot be made on the basis of measuring CO_2 leaving the chimney.

The basic attitude to actual long-term analysis is that the economic inflow, by chemical forming and by mobilisation of stocks, and the economic outflow, by chemical destruction and immobilisation of carbon, are of prime importance. Changes in the speed of some parts of the biological carbon cycle, e.g. through harvesting more wood or letting it grow somewhat longer, may be important in the short term. They do not increase or diminish the total amount of carbon in the cycle. *In finding solutions the central aim here is to*

[1] It is excluded here for practical reasons. The amounts are insignificant and most soda comes from natural sources.

limit the human contributions to the total amount of carbon in the biological cycle by controlling structural inflow and structural outflow.

Discussion of results

Several features are remarkable. As expected, the structural human contribution to the carbon cycle is dominated by the extraction and combustion of fossil fuels. Chemical forming does not contribute more than two and a half percent. The part of the biological cycle itself that passes through the economy is dominated by wood. The harvest of wood as a product and through deforestation covers seventy-five percent of total inflow from the environment. The food share, for direct human consumption, takes only a small fraction of total primary production in agriculture, although covering large areas. Animal husbandry takes a large portion of food and fodder carbon, which is then mainly emitted as respiration and manure. Direct human consumption, including meat, is a minor item.

At the outflow side of the economy, there are some peculiar facts. First, wood used in stable applications, as a building material and as furniture, might give a substantial contribution to outflow to the lithosphere. This is a specific human contribution to structural outflow, comparable to peat forming in the environment. Secondly, it is curious that the potential for structural outflow is not used at all, neither high-tech disposal of CO_2 in gas domes or in clathrates at the bottom of the deep seas, nor easy solutions in the form of permanent storage of discarded wood and other organic materials in waste. Technically, such options might become operational at short notice. Current developments towards more waste burning, with low efficiency in energy recovery, are clearly in the wrong direction as far as CO_2 is concerned. The third and most remarkable feature, however, is the preponderance of methane in global warming[1]. It constitutes over half of all human GWP_{100} contributions. Spills of methane at coal mining and at methane gas distribution are main sources. Ruminants take second place and paddy fields third. No contribution of sewer and manure to CH_4 emissions has yet been assumed.

Methane contribution is already substantial using the long-term GWP_{100} of 21 stipulated by the IPCC[2]. For a shorter time horizon of two decades this is the case even more. The twenty year GWP_{20} of methane is 63, according to the IPCC. More recent computation shows slightly lower figures. Lelieveld and Crutzen (1991) arrive at 51.4. *Current contributions to global warming in the next decades will not be primarily due to CO_2 related emissions.* They will be primarily due to methane, which will account for about sixty percent[3].

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[1] This figure is substantially higher for CH_4 than that given by the IPCC, see the comparison, also with other sources, in Appendix 3. The question marks in table 5.5.2 indicate that my figure still might be a lower estimate. If this figure of methane emissions from current biological processes goes up, the figure for geological methane would rise proportionally. However, the modelling of the breakdown of methane would have to be adjusted as well, to make the emissions commensurate with actual concentrations. Then both the GWP_{100} and the GWP_{20} of methane would become smaller.

[2] The seemingly much lower contribution indicated in the IPCC data in Table 5.2.2 is based on the fact that IPCC uses gross emissions while here percentages are based on net emissions.

[3] The choice of two periods of GWP analysis customarily used are 20 years and 100 years. This seems a mixed choice. More relevant would be one measure that is independent of time, through the integration of the effect in time, that is GWP_∞. The other would be the effect induced in the short term, e.g. next year, as GWP_1. An indication of the timing of integrated effects could best be based on the time that half the climate forcing is realised, the GWP-*lifetime*. The time independent GWP_∞ of methane is around 8, the short-term GWP is well above 100, in the range of 10^3.

How reliable are the results of our preliminary analysis? There are some gaps and some boundary problems that may influence results. The gaps in the economic flows related to CO_2 are probably minor. Only the amounts related to agricultural erosion are based on unverifiable assumptions. The main gaps relate to methane. The nett methane forming from current biological carbon flows probably is an underestimate. Emissions from *manure, fermentation in new waste biomass, fermentation in old biomass, as from deep ploughing* have been disregarded.

TABLE 5.5.4 CONTRIBUTION OF CO_2 AND CH_4 TO NETT TOTAL GLOBAL WARMING IN GWP_{100} AND GWP_{20}[1] IN 10^9 UNITS GWP, AND SHARES IN PERCENTAGES. SOURCE: APPENDIX 5.5.3

-Combustion processes		$GWP_{100/20}$ CO_2	GWP_{100} CH_4	GWP_{100} CH_4%	GWP_{20} CH_4	GWP_{20} CH_4%
*fossil fuels		21805	?	?	?	?
*wood		2427	560	1.9%	1680	3.5%
*wastes	plastics	4	?	?	?	?
	organic	62	?	?	?	?
-Biotic processes						
*from ruminants directly		0	1680	5.7%	5040	10.4%
*rice growing in paddy fields		0	1260	4.3%	3780	7.8%
*organic materials (landfill)		0	1260	4.3%	3780	7.8%
-Chemical forming						
*cement		450	0	0.0%	0	0.0%
*plasters		88	0	0.0%	0	0.0%
*industrial. applic.		4	0	0.0%	0	0.0%
*oxidation of carbon in steel		0	0	0.0%	0	0.0%
-Organic materials to environment						
*biomass, faeces+hum. respir.		253	?	?	?	?
manure+anim. respir.		2234	?	?	?	?
*pre-production losses						
oil		88	0	0.0%	0	0.0%
coal		23	0	0.0%	0	0.0%
*fossil methane spills		0	3304	11.2%	9912	20.5%
-Net oxidation of agric. soils						
*erosion		953	?	?	?	?
-Deforestation		5683	1400	4.8%	4200	8.7%
Gross GWP	*by CO_2*	*34137*				
caused	*by CH_4*		*9464*	*32.2%*	*28392*	*58.7%*
GWP extracted	by CO_2	14212				
	by CH_4		0	0.0%		0.0%
Net GWP	**by CO_2**	**19925**		**(CO_2: 67.8%)**		**(CO_2: 41.3%)**
caused	**by CH_4**		**9464**	**32.2%**	**28392**	**58.7%**
	by total carbon	**29389**		**100.0%**	**48317**	**100.0%**

[1] GWP-100 of methane is 21. Methane mass is 16/12 times C-methane mass. Carbon dioxide mass is 44/12 times C-carbondioxide mass.

If the estimate of methane emissions from current biological sources would go up, the estimate of fossil methane contributions (both man-made and natural) rises proportionally. C_{14} concentrations may give a clear indication of the relative amounts of fossil methane and new methane. The half-life of C_{14} is 5586 years. The amounts are clear only if just these two categories are concerned, old without C_{14}, and new with known amounts of C_{14}. However, there is also a reflux from carbon stocks more recently deposited, both from natural sources and as man-induced flows. The man-induced reflux with a variable time of storage might be substantial. Agricultural erosion brings deposits to the fore that may contain organic carbon formed hundreds to thousands of years ago. The ratio then is from new to old plus not-so-old. This may correspond to a highly variable amount of methane. Whatever the outcome of the new estimates will be, the importance of methane in terms of relative contribution in global warming will remain quite constant. This is due to the corresponding change in its changed global warming potential the also resulting. Thus current data give a reasonable data-independent estimate of the importance of methane in global warming.

Boundary problems relate mainly to agriculture and forestry. A somewhat ambiguous position seemed best. CO_2 related streams in agriculture that remain in the fields have been disregarded in the analysis of economic flows. It comprises roots, stubs, branches, leaves that are not used commercially. If, however, these organic materials decompose, giving off methane, as in rice paddy fields, these CH_4 emissions have been registered as emissions from the economy[1]. Adding the non-commercial parts of agricultural production to economic flows would lead to a very substantial addition to economic inflow and a very substantial addition to economic outflow as well. If techniques to influence the manner of outflow were to become available, inclusion would be necessary. Now, excluding them seems most practical.

5.5.5 Reducing climate changing emissions: technical solutions

In the search for reductions of global warming causing emissions, all possibilities should be scanned and treated similarly, ultimately in terms of their effect on nett emissions. The IPCC gives a scenario to stabilize concentrations of climate gases at current levels[2]. CO_2 emission then should drop by more than sixty percent, those of methane only by fifteen to twenty percent. As they themselves point out, it seems rather odd to take historical values as a norm. It makes more sense to specify the level of GWP contributions deemed acceptable. Reductions in GWP then could be effected in the most efficient manner, taking into account all substances and all sources. Given some cost restraints, effectiveness in terms of 'climate change prevention' would be created in the latter approach.

For practical reasons[3], only carbon dioxide and precursors, and methane have been singled out for policy development. The analysis is *based on their long-term contributions only, as GWP_{100}*. Reductions of nett CO_2 and CH_4 emissions are possible in several ways.

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[1] A corresponding carbon inflow has been added in the data for balancing reasons.
[2] Houghton 1990, p.xvii.
[3] My limited capabilities.

They may be grouped according to the way they reduce the human contribution to the global carbon cycle. The main categories are

*1. reduced inflow from lithosphere
*2. reduced chemical forming
*3. increased outflow to lithosphere
*4. increased chemical destruction
*5. increased accumulation in the economy
*6. regulating mechanisms in the environment.

It should again be noted that there is not much place in this list for such very sensible measures as not destroying forests. They help in the short term, but not so much in the long term.

1. Reduced inflow from immobile stocks

Direct reduction of inflow is possible if, first, the function of products is not altered, for better or worse, by the change. Which substitutions could function so independently from the further working of society? Let us try an example. Current organic chemistry uses about two percent of carbon in oil in polymers. This carbon use might be replaced to a large extent by silicon-based chemistry. Process energy need not change very much. Carbon for materials would be saved. Usually, however, costs are different and functional characteristics change substantially. In that case such direct technical options have their place only in a larger socio-economic setting, with indirect and secondary reactions in society taken into account. Secondly however, the change in carbon use should not have any influence on the market for fossil resources. This clearly is impossible, a reduction in demand will lower prices and will lead to increases in other applications of, e.g., oil. Thus all techniques for inflow reduction involve the broader functioning of society. They all have some technical kernel placed in a broader societal setting, at least encompassing economic relations of supply and demand.

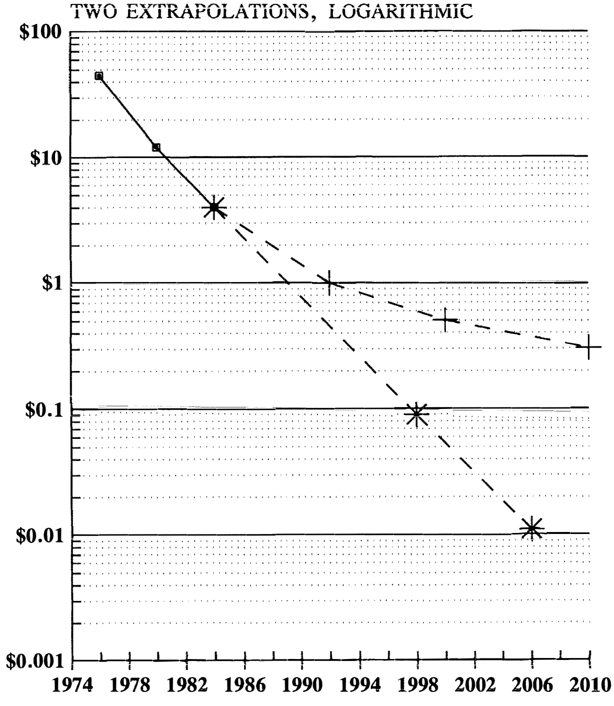
Process integrated improvements

Which process integrated improvements may be designed? This is a very broad field. One option is improving energy efficiency of combustion processes. Such developments often require large amounts of R & D. Ceramic pistons for car engines are an example. Another option is to use waste heat for economic purposes. A wide range of options are open that often are attractive financially even at current prices. One reason for not using these options are institutional barriers. In many countries the monopolistic suppliers of electricity allow private combined cycle production of heat and electricity only under unattractive conditions. See Tellegen et al. (1991, p.49) for institutional solutions. The nett effect, through indirect and secondary mechanisms, amount to less than direct effects. The reduced oil demand from the combined cycle plant will lower prices and induce others to consume more. The nett effect depends on the long-term elasticity of supply and demand for fossil energy. As the demand curve shifts downward, there will be some price drop and some diminished use. The nett effect on carbon inflow is undisputed in its direction, not in its amounts. Direct efficiency increases similarly reduce the inflow of carbon from fossil fuels, with the indirect effect lower than the direct one.

Non-fossil energy resources

Secondly, other sources of energy may be used that do not contain carbon. Possibilities are nuclear power, all sorts of direct and indirect solar power, and earth heat. The costs of these options are not clear. Nuclear power from uranium seems financially attractive. However, if strict and several liability were to be established, without upper limits and without evading constructions, the costs of nuclear power would certainly rise. The costs of electricity from breeder reactors, currently much more expensive than traditional nuclear power, would rise even more. In the long run the solar cells seem the most attractive options. Their capital costs show a steady decline over the last decades, see 5.5.3. The decline in cost price per unit installed dropped by over twenty percent per year from 1976 to 1984. If this trend were to continue, capital costs in primary electricity production would drop to insignificant levels within a decade from now. See the irresponsibly optimistic lower striped line in figure 5.5.4. However, much less optimistic assumptions on technical progress will have the same effect, see the line with crosses, only in decades. See the upper extrapolation where technical progress drops from twenty-one percent to seventeen, to nine percent, and finally to seven percent after 2000.

FIGURE 5.5.4 COSTS OF PHOTOVOLTAIC POWER, $ PER PEAK WATT INSTALLED[1], WITH TWO EXTRAPOLATIONS, LOGARITHMIC



Hamakawa, in 1987, predicted a cost prices per kilowatt-hour of about $0.08 by 1995, based on a projection somewhere between the striped predictions. One Californian research group already claimed in 1991 that it had developed cadmium sulfide cells that produce electricity for $0.08 per kWh. Current cost prices of fossil energy based electricity are in the order of $0.06. If the claims are true, one or two more years should make solar cell generally competitive. However, solar cells that produce electricity have

[1] Source: Hamakawa (1987)

one serious drawback: their production varies with the amount of radiation they receive, both per day and per season, and is also irregular because of weather conditions. Energy storing facilities are a requirement for such solar cells. Producing hydrogen instead of electricity does not have this disadvantage. Hydrogen may be stored, it may be transformed into hydrocarbons and it may be transported over long distances[1]. Large-scale power production from hydrogen is possible with relatively low capital costs. Small-scale power production is possible with a high efficiency. For mobile use, costly small-scale storage of hydrogen is required. Costs are difficult to quantify. Even if the overall costs of hydrogen production are lower than current energy systems their large-scale application may be retarded by the fact that a systems change requires coordinated decisions at different places in society. Forced system development may be effected by regulations as for cars in California. There, by the year 2000, producers and importers of cars will have to sell ten percent of all cars emission free. Electric cars are the most probable option. Also, if a change occurs, effects on inflow, outflow, and emissions are dampened here by economic mechanisms. Producers of fossil fuels, e.g., will lower their prices, before they lose their markets.

Total carbon efficiency by recycling

Thirdly, overall energy/carbon efficiency of production-consumption systems may be improved by recycling of products and materials. If recycling of aluminium increases, the primary production goes down at a given level of use of aluminium, as will total energy consumption for aluminium. This will be true for aluminium since the primary production of this material requires so much energy. It is not so clear whether carbon emissions will drop as well. Aluminium is usually produced with hydro-electric power and earth heat (as in Iceland), without carbon emissions. There might easily be a nett increase of carbon emissions since recycling requires transport, heating and cleaning of the aluminium in places where hydro-power is not available. Recycling would then increase carbon emissions. However, in a more complex model of society, the growing possibilities for substitution between hydro-power and fossil fuels may be taken into account. In that case part of the hydro-power may well replace fossil fuel. In such a more complicated analysis, contrary to results form a simple analysis, recycling of aluminium might seem to contribute again to diminished emissions of CO_2. For other materials this relation is even less clear. Recycling of paper requires energy. In an energy efficient society, waste treatment of paper would lead to the recovery of the energy content of the paper as through electricity production at combustion, or through methane recovery by active fermentation or by fermentation from landfill. The nett effect of recycling on energy consumption might well be negative, assuming that primary production of paper requires less energy then is recovered at final waste handling. This is even more the case for CO_2 emissions since the electricity generated at incineration replaces electricity produced mainly with fossil fuels. With recycling, secondary electricity production from tree based paper goes down. Again, using a more complex model of society may change the picture. Suppose the long-term supply of wood for pulp production is inelastic. The wood not needed in case of recycling will find an alternative application as paper. In the long run there will be the same amount of paper burned. Part of the paper will be recycled first as

[1] There are many systems for storing solar energy, based on primary electricity production or on chemical reactions. See Dostrovsky (1991) for a survey.

paper, increasing the total use of paper, and part will go into other applications, partly replacing other materials like textiles and plastics. The extra use of recycled paper would increase carbon emissions through the energy required for the paper producing recycling process. Other applications would save other materials, probably leading to a nett saving on energy and CO_2 emissions for these applications. The nett effect depends on the precise quantitative relations modelled. It could be positive or negative.

Now suppose that the supply of pulp wood is elastic. With increased recycling, total virgin paper production will decrease, as could wood production. Recycling then could easily lead to an increased contribution to global warming. Recycling is an option in all cases where the substance studied has a valued function per se. With carbon it usually is not the carbon that has a value[1] but the potential energy that is contained in its chemical bonds. If this potential energy is used as actual energy it is lost and the carbon has no value left in this respect. Such recycling of carbon therefore is never an option in reducing CO_2 emissions. In all other cases of recycling, modelled predictions of a quite complicated nature are required for assessing nett effects. They have to represent the actual complexities of indirect and secondary effects. There are few methods available for developing the relevant models. Policies now directed at these types of options cannot be based firmly on an unequivocal analysis of expected effects, not even on the direction of these effects.

Low-energy consumption
Fourthly, less final use of energy requiring consumptive activities lowers the inflow of carbon. Some small contributions might be quite cheap. In many public buildings a lower in-door temperature in winter and a higher one in summer are often more pleasant and always less costly. In addition, such measures reduce the use of fossil fuels, if these are used for heating. Reduction of the general volume of high energy activities would be very costly if this reduction is to be substantial. Energy seems a booster of economic growth especially in countries with an average income somewhat lower than the current OECD levels. The costs of substantial volume reductions are probably higher than those of the other types of options. They also depend on the scale on which these options are introduced. The costs of volume reductions of energy-demanding activities are not influenced by such technological changes as a shift to other non-fossil sources of energy. After such shifts, the emission reduction of volume changes, however, becomes smaller. If the proportion of non-fossil fuels rises, the costs of volume reductions will thus rise as well, per unit of emission reduction. Initially, lowering the final output of some commodity will decrease production, resulting in the desired positive effect on carbon inflow. Demand may shift directly to other activities. If not unemployment will result. The pressure towards full employment will result in macro-economic policies that will replace the former demand for high-energy products and activities. The nett effect will probably be a tiny fraction only of the direct one.

All four types of options may be seen from a product or process oriented point of view. In each case, effectiveness may be computed as the contribution of only a single change.

[1] Exceptions are all chemicals and all non-energy foods, and perhaps some very small applications such as diamonds and fibres in composites. Their share in total carbon flows is insignificant.

One product replaces another functionally equivalent product, or one process replaces another process with the same function. For the economy as a whole, other mechanisms may subtract, and sometimes add, to the effectiveness of this type of micro analysis. If the supply of fossil energy is totally inelastic, reduced requirements for specific applications will lead to such price decreases that demand from other applications will absorb the full amount produced. At present, long-term supply is never totally inelastic. nett reductions in total inflow will therefore not be zero. They will, however, only be a fraction of the inflow reduction realized by one specific option. This reduced effectiveness can also be expressed in terms of costs and prices. If solar cells are to receive a substantial share in electricity production their cost-advantage must remain after the price change induced by taking only a small market share. This means that if the coming generation of solar cells will be competitive at current prices in the near future they not yet acquire a large share of electricity generation. Combined in a realistically complex model with the indirect and secondary effects of most options, the nett effects will usually be much smaller than the direct effects. Some product and process changes may even be counterproductive. Only increased energy efficiency and energy substitution work for sure, albeit at a much lower level than direct effects would indicate. Increasing complexity of the models that predict effects will usually dampen the nett effect. Other inflow reducing measures, such as recycling and changed types of consumption, cannot form a well-founded basis for the reduction of carbon inflow. Their potential contribution might be very large, small, or even negative.

Some options may be attractive because of their combined contribution to the reduction of several environmental problems. A shift from private car kilometres to public transport car kilometres reduces the emissions of several hazardous substances besides CO_2, the number of roads required, traffic noise, accidents and the mining of several other non-energy resources. In such an analysis, however, we leave the realm of substance flow analysis and change over to the life cycle analysis of products. The combination of these two modes of analysis gives the full reality. That however, is too complex to analyze, now or in any foreseeable future.

2. Reduced chemical forming in the economy

Diminished industrial transformation of carbonates into CO_2 is the major option. This may be realized by reducing the production of cement, or by using a carbon-free source of calcium such as gypsum[1]. Plasters may be partly or fully based on gypsum. Cement from steel production is similarly formed from limestone. Some limestone is always required at steel production to keep slag manageable. Reducing the production of steel, cement and plaster has a positive primary effect. Secondary effects are dampened because the functions of cement and steel will partly be taken over by fossil based plastics and by ceramics that may require even more energy.

Next, upon incineration all carbonates in waste will burn to CO_2, as is also the case with cremation. The analysis of carbonate incineration is completely mixed with the energy part of organic wastes. Incineration will usually co-produce some electricity with, *on average,* a low efficiency. This low efficiency is due to non-burning parts of waste,

[1] This would change the hardening properties of cement, perhaps to an unacceptable degree. It also would free the sulfur in gypsum, leading to extra SO_2 emissions.

moisture and the less effective technical facilities for burning waste as compared to furnaces for coal and gas. Per unit of electricity, prime fossil energy use, combined with the stable permanent storage of waste, would have lower carbon emissions than incineration. Indirect effects would here reinforce the prime effect. Higher demand leads to higher prices, thus pushing other demand off the market. If a choice were to be made between incineration and stable storage, at the same costs, the latter would always be preferable because of fewer CO_2 emissions. However, stable storage is an ideal that will never be fully realised.

3. Increased outflow to lithosphere

Increased outflow to the lithosphere may be realized with end-of-pipe solutions. These do not much change the inflow of carbon into the economy but they do affect the way it flows out of the economy. Several techniques are available. CO_2 might be stored in many ways. It can be stored in the sites of used-up natural gas reserves. Currently, this seems the cheapest solution. The costs of this option have been computed by Shell in a study for the Dutch Ministry of the Environment. The case investigated was a coal gasification and electricity generating plant in Buggenum, the Netherlands, with gas stored in a depleted natural gas dome one hundred kilometres away. nett total costs amount to ECU 0.073 per kilogram carbon stored[1]. CO_2 may also be deposited in the deep oceans, a technique which would remove it from the carbon cycle for a very long time. At high pressures and low temperatures the CO_2 may be trapped in the form of clathrates. Similar to the clathrate storage of natural gas, these are ice-like substances that may be deposited in a solid form. The long-term stability of these compounds is not yet fully clear. Even if they would slowly release the CO_2 thus deposited in the deep oceans, it would still be a good long-term option, though maybe not "back to the lithosphere". The surfacing of deep ocean water takes about a thousand years, in a cycle in-between the biological and geological one. The amount of CO_2 in the oceans is about 50 times as high as that in the atmosphere. In the long, to extremely long term only about two percent of the CO_2 thus deposited would contribute to global warming[2].

Organically bound carbon may effectively be taken out of the carbon cycle as well. Isolating the more stable forms, as with burying PVC pipes, is a primary option here. If organic wastes from households and sewers could be deposited in a way that stored the carbon permanently, a substantial contribution to diminished emissions could be made. Two options seem available. One is to dry the waste. Research on old waste dump sites in the US indicates that such storage may last a very long time. In a site in Arizona a journal of over half a century old that was unearthed could be read like yesterday's paper, see Rathje and Murphy (1992). The other option is to store the waste oxygen free, underwater. Some material would decompose anaerobically, giving off methane to be recovered. The remainder could be left alone indefinitely. Placed in a stable geological setting the waste would become peat, lignite and finally coal, all of poor quality because

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1 Partially offsetting costs is the extra production of natural gas from the field used, through the build-up of pressure. This extra supply has not been subtracted. In the US Fluor-Daniel has installed a system that takes CO_2 from electricity production and introduces it in oil wells for Enhanced Oil Recovery, with a capacity of over 1000 tonnes a day.

2 The limited solubility of CO_2 at atmospheric pressure would lead to a partition coefficient with more CO_2 reaching the atmosphere. The pH buffering of the oceans would lead to the binding of CO_2, effectively withdrawing CO_2 from circulation.

of the low carbon content. This option is very similar to waste storage in the economy, see below. The latter option requires permanent activities to preserve the dump in the manner required.

4. Increased chemical destruction
Chemical destruction can take place by binding carbon irreversibly, which under normal conditions is in the economy and the environment. It is then removed effectively from the carbon cycle. All insoluble carbonates, salts of carbonic acid with metals can have that function. Gypsum may be transformed into limestone, while sulfur is produced at the same time. This is the reversed of the process used in the cleaning of sulfur from combustion gasses. Many compounds of carbon exist with similar or even stronger bonds. Carborundum (silicon carbide, SiC) is an extreme example of such stable compounds, as is diamond. Another feature of chemical destruction is that economic mechanisms reduce nett effectiveness. If carborundum is produced as a useful material, the demand for carbon rises and so does its price. More of it will be produced, though not the full amount then used in carborundum. The material's carbon which is taken out of circulation does not contribute to global warming but neither did the oil embodied in it when it as still in the ground. Only the part of other carbon applications displaced contributes to nett emission reduction.

5. Accumulation in the economy
Accumulation in the economy gives only temporary relief. For such an urgent problem as climate change, temporary solutions may still be important, and "temporary" may be very long, transforming into "back to substrate" in due time. The time horizon of accumulation in the economy depends on the type of application. It may last up to centuries in the construction of buildings. The accumulation may concern both the carbon put into the economy from fossil resources and the carbon extracted from the environment as biomass. The accumulation of carbon harvested from nature has the greater potential. Biodegradation is the largest threat to long-term accumulation. Most plastics are much less biodegradable than biomass from the environment. Even discarded food and fibres could be stored for quite a long period. Technically this is very similar to accumulation in immobile stocks, see above. For most waste streams from households dry conditions or anaerobic wet conditions would be sufficient for extremely long-term storage. Not burning organic wastes but conserving them in dump sites could contribute substantially, several percent points, in reducing current emissions. See van Duin et al. (1991) for quantification concerning plastic waste[1].

6. Regulating the environment
Some measures may work only by influencing the environment. Drying marshes will lead to extra oxidation of peat into CO_2 and other compounds. Induced peat formation is one of the possibilities for immobilizing carbon. This option is quite unlike forestation and deforestation, which is a change in biological flows and not in geological flows. The peat forming option may also be interesting because it supports the conservationist efforts to

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[1] Their theme is CO_2 emission reduction through waste policies. They quantify the combined effect of not burning, and thus not recovering energy, with recycling options that reduce the need for prime materials. The latter effects are dampened by supply and demand effects. They do not quantify the much larger amounts of natural organic materials that may be taken out of the carbon cycle.

save the last peat bogs in Western Europe. This type of option does not fit well into the substance flow scheme. It is regulating processes in the environment in such a way that carbon is immobilized in the environment. If regulating the biological environment and its geological relations to the substrate dominates environmental processes for other substances as well as carbon, the distinction between economy and environment will become fuzzy. My sharp distinction between economy and environment will then lose its meaning. The biotic environment would become a combined zoo and garden, with some fish keeping and animal husbandry integrated into it[1].

Indirectly, non-carbon emissions will influence processes in the environment. Emitting acids will lead to the increased weathering of lime. Emissions of SO_x, for example, transform limestone into gypsum and CO_2. Sulfur is a "global cooling gas"; it has a negative Global Warming Potential. Through weathering, it makes a positive contribution to global warming. This indirect effect implies a positive element in its Global Warming Potential[2]. The cooling effect covers a very short period of time, the warming effect lasts very long. Acids have not yet been included in lists of GWPs.

There are many ways to slow down or increase the speed of processes in the global carbon cycle, such as (re)forestation and the spreading of iron and silicon in the oceans. Whatever their merits, they do not belong to the analysis here since they relate to short-term changes in the carbon cycle. Some combinations seem attractive, or not. When changing to hydro-electricity in newly formed reservoirs, removes the former vegetation effectively. In one study is has been found that for fifty years emissions result, of CO_2 and methane that are similar per kilowatt-hour to those associated with coal based electricity[3]. In this analysis a biological flow and a structural flow are mixed. Similarly, electricity utilities in Europe "compensate" their emission by reforestation projects. Such mixes are not sensible in designing instruments and instrument strategies for long term policy. If one includes biological flows in the analysis, this should be done systematically.

Methane
For methane, the options are very different. The largest source is associated with the extraction of fossil fuels, a large share taken by coal mining. Technically, these losses may be reduced substantially, as is the case of distribution losses of methane. Techniques are contemplated already for safety reasons of first removing the methane through vents before the actual coal mining starts. The second largest source is ruminants. Eating less meat and drinking less milk would help. The third largest source is rice in paddy fields. Reductions there would reduce food production, or at least the (subjective) quality of alternative food produced. Finally, there is the anaerobic breakdown of organic material in the economy, as yet only partly investigated. The solution there could be to either burn

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1 The Dutch State Institute for Research of the Sea (NIOZ) warned that the reduced use of phosphates in washing powder would have "serious repercussions for the amount of fish to be caught from the North Sea". Transforming all environmental processes into economic processes may be a serious long-term option. The question is whether overall control over such a complicated system can be stable enough in time to avoid mass extinction and irregular decimation of human populations. And of course, serious side effects may make such short-term options unattractive. Algal blooming may be good for transport of carbon to the deeper seas. Some algae may kill the neighbouring fish and shellfish in the process.

2 Secondary effects are usually included in computing Global Warming Potentials of substances.

3 Report in New Scientist of 24 July 1993, p.11, on a study on Cedar Lake reservoir in Manitoba.

the methane, as increased outflow after collection, or prevent the methane from forming, thus reducing its inflow.

Discussion
The very broad applications of carbon compounds, in a vast number of different functions, make it extremely difficult to define techniques for emission reduction. Only end-of-pipe techniques work for certain in amounts coinciding with the techniques specified. All other techniques are so much embedded in economic and social processes that changes in one place may be partly or even fully offset by induced changes in other processes. The complexity of the model used determines the level of effects much more than the characteristics of the technique analysed. A general feature of the technical solutions is that their micro analysis, of the change from one process to another, may yield clear and substantial results. Increasing the aggregation of models used to predict nett effects will lead to dampened and sometimes even reversed effects, as with some recycling measures.

At the same time, the extreme variability of all related processes indicates that very great changes are indeed possible if proper steering mechanisms are developed. For large emission reductions, combined with a high level of energy use, either or both of two main technical approaches are necessary ingredients. First, increased outflow to immobile stocks may become a very important option for CO_2. Storage of CO_2 has a large potential. It is the only technique for which a first reasonable estimate of costs is available now. Storage of waste biomass, especially wood, might contribute substantially at little cost by fixing an amount of carbon equivalent of ten percent of fossil inflow. Or, secondly, a change over from fossil energy to other sources of energy must take place. Which techniques are most favourable overall is not clear. Electricity and hydrogen from solar cells in deserts seem to be the best long-term options. For methane a number of technical measures may lead to substantial reductions even in the short term.

5.5.6 Reducing climate changing emissions: policy instruments

Which policy instruments are available for reducing emissions of CO_2 and CH_4, and for emissions of CO_2 and CH_4 forming substances[1]? Which context is assumed for their implementation? The large difference in the long-term global warming potential between CO_2 and CH_4, by a factor of 21, requires a separate handling of these two substances. Policy instruments will have to differentiate between the two substances. Even if instruments may be applied similarly to each substance, the level of the tax required might still indicate different choices to be made in the flexible response strategy. Not all emissions are relevant for structural policy. Only those flows have to be taken into account here that add to, or subtract from the biological carbon cycle. Carbonates emitted at cement production, for example, are relevant because they derive from a carbon source not taking part in the biological carbon cycle before. Emissions from the burning of currently produced biomass are not relevant in the long run since they are directly based on the forming of the biomass. Not-emitting them, through long-term storage, would make a difference. CO_2 emissions from waste incineration are partly relevant, for the part

[1] It is advisable to make the other greenhouse gases the subject of policy development at the same time. Otherwise large-scale problem-shifting might occur, as has been indicated several times before. See Lelieveld and Crutzen 1991 and van de Vate 1991 in this sense.

deriving from fossil carbon resources. Instruments should cover all these widely diverging options.

No specific administrative level is assumed. Generally, it is most attractive to treat environmental problems at the level at which they manifest themselves. Then policies are not weakened by the free-rider behaviour of other administrative units. For the global problem of climate change there is no corresponding global administrative level for policy development and implementation. Through treaties, governments may bind themselves to certain aims and practices. These treaties themselves may become instruments for behavioral regulation, not of emitters but of governments[1]. Through which policy instruments the aims are realised within a country remains a matter of free choice for each individual government. The policy strategy in this case study does not assume country-specific modes of implementation. Supposedly, a taxing apparatus of reasonable quality exists in the form of duty and excise offices and some form of environmental agency has the capacity to classify and measure environmentally hazardous flows. This is the case in all industrialised countries. For reasons of exposition and simplification the world is treated here as one administrative unit. Problem-shifting to other countries, a very complicated factor in the analysis of individual countries, thus cannot occur. Of course, actual implementation at only some lower administrative levels will be more complex and more costly. This section lists instruments applicable to several streams. The list treats main types of instruments in the by now familiar descending order of freedom and efficiency. The list does *not* give all instruments applicable to all flows. Only the more macro instruments with a reasonable chance to fit into the flexible response strategy are on the list. If a flow can be managed easily, e.g. under the first-choice deposit scheme, it makes no sense to look for second-choice options. Nevertheless, some flows might be handled by different instruments. Based on the list that is applicable to the two types of carbon flows, the next section describes the flexible response strategy as a choice from the list, starting from the top until all emissions are covered. Finally, a rough approximation of results to be expected is given, based on a technical scenario.

The preference order of instruments, again, is:

Structural instruments
- (Extended liability)

Cultural instruments
- Standard methodology for LCA

Financial instruments
- Substance deposit
- Uniform emission tax
- Estimated emission tax

Prohibiting instruments
- General emission design standard
- Estimated emission design standard

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[1] The Montreal Convention and its additions state aims in terms of reductions of amounts produced and emitted of certain ozone depleting substances for each country. Such a system might be replaced by a system of permits that are tradable between countries. For a given amount of allowable national emissions governments then are still free to choose any instrument - from direct regulations to persuasion - to reach the emission level for which it has permits. Of course, the treaty might also state that a system for tradable emission rights is the instrument to use at a national level to actually realize the emission reductions required. Such international agreements seem highly unlikely politically. See OECD (1991b), where the distinction between regulating governments and regulating emitters is not made so clearly.

Structural instruments

The general principles of property, especially the liability rules protecting it, might be extended to cover damages because of global warming. Strict and several liability is required for a reasonable chance of application to environmental problems. Of all environmental problems, liability rules are least applicable to the climate problem, if at all. There are several factors that make liability inapplicable. First, causation in the emission of carbon compounds is so diffuse that it is quite impossible to single out specific persons or organizations to sue as responsible. Nearly all firms and all people on earth contribute to the climate problem directly.

Secondly, the problem itself is a prime example of a collective good that is endangered, giving no one a special right to claim damages. Climate changes affect all people on earth. For some people, however, the consequences might be more serious than for others.

Thirdly, the time horizon of effects is extremely long. A molecule of CO_2 emitted now may exert its heat retaining effect for several centuries.

For these reasons, even extended liability rules cannot contribute to emission reductions of CO_2[1]. They might even lead to extra emissions. The costs of nuclear energy will rise relatively because risks are attributable and extremely high, remember Chernobyl[2]. Raising the costs of nuclear energy, will rightly decrease its share in total energy production, leading to an increased use of fossil fuels and other types of energy, all as compared to an assumed autonomous development.

Some changes in laws on landed property could have rapid effects, at the borderline between the biological and the geological carbon cycle. Some laws make the destruction of forests economically very attractive. In Latin America and in less so in Southeast Asia the laws on acquiring land usually state that he is the owner who actually works the fields for a given period. This period ranges from seven years (e.g. French Guyana) to ten years (e.g. Costa Rica). Before that date the 'provisional property', may be sold to someone who then becomes the full owner. If the new owner does not work the fields, a squatter may again take the property and, through actual possession, become the new owner later on. Such rules might even be applied against the landed property of large landowners and public authorities. See Dryzek (1987) on how this system works in Costa Rica, a relatively enlightened country spending substantial amounts of money and effort on protecting its tropical rainforests. Even there, however, removing trees from an area and putting some cattle on it will make you the owner of the land in due time. If the wood cannot be sold by lack of transport facilities this mechanism still leads to the stripping of forests even in sparsely populated areas. This happens at the eastern side of the Andes, e.g. in Ecuador[3]. Expectations of future developments may be reason enough

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[1] The tax on sulfur in Japan was payment in a liability procedure through administrative law. Proceeds were paid to victims of air pollution with respiratory problems. There current emitters paid for current health problems. This method might be applied when damages of climate problem occur. Effectively, there is then an emission tax, with proceeds used in a certain manner. The other reasons stated would still make application impossible.

[2] If strict and several liability had been applicable and had effectively been applied there, the former Soviet, now Ukrainian, utilities would have received the largest ever bill in history. It may be doubted if under such circumstances newer installations would have been commissioned.

[3] Personal communication Huber, project leader of a project in Ecuador that investigates factors contributing to the decline of tropical forests.

to burn[1] down the forest. If someone else already starts logging and burning, it is too late for you to earn the large profits that sale then might bring. Pioneers, often creating their own law and order by force, may be motivated more by the chance of large amounts of money than by the long-term proceeds of their agricultural activities. As long as current proceeds cover current costs they have a stake in the lottery of possible future development. If after some time the soil becomes exhausted and erodes, nothing has been lost but the lottery ticket. A new one can be acquired easily by logging and burning the next area. Of course, in many instances good farmers may transform forests into permanent agricultural lands or pasture. Good farmers, educated for the difficult task of land reclamation and of the development of modern tropical agriculture, are a much scarcer personality type in the tropics than poor adventurers trying to become rich the easy way. Supporting the adventurers, as current laws do, seems bad for climate, bad for nature, and bad for real economic development as well.

Systems for acquiring landed property in most Western countries have fully registered land ownership in public land registers, with strict procedural rules on change of ownership that are not related to actual possession. Introduction of such rules would eliminate one mechanism, amongst others, that now leads to deforestation and desertification. Exclusive land ownership does not exclude public ownership. It excludes non-ownership. However, public ownership still requires public activities to regulate the use, or non-use, of public lands. One of the largest short-term improvements by not (yet) logging tropical forests, could be supported by a change in ownership rules. This type of legal instrument would not influence structural inflow and outflow very much, being related to biological flows primarily.

Cultural instruments: first and second choice

There are no first-choice cultural instruments for the climate problem. Substance related information can only be relevant if it happens to coincide fully with product information, based on life cycle analysis (LCA). Even in the conversion of fossil energy, however, there are many more harmful substances than those which are carbon related, and many more environmental problems than that of climate. Information on products and processes can thus play only a limited role. There might be a more prominent but still second-choice role in the diffusion of existing energy saving techniques. In all other instances, and even there to some extent, other detrimental environmental effects have to be included in the information to arrive at a balanced environmental view. Information on processes may be of the same type as product LCAs since processes with a similar function are compared. In the analysis of specific processes of individual firms, environmental audits will be the most apt instrument for assessment. Only in special cases can the substance flow analysis for just one environmental problem play a central role in these LCAs or audits. Such an exceptional example is the retail distribution of methane, see Wit, Taselaar, Heijungs and Huppes (1993). A comparison between utilities in terms of percentages lost might be a strong motive for improvement, especially as the consumer is forced to pay the losses.

[1] Rumours that coca farming may become a "protected" activity in some location could be an important factor now in the deforestation of the less accessible rainforests in South America.

Cultural instruments: third choice
Publication of the names of large emitters is practised now in the US. For CO_2 the large emitters would be refineries, electricity utilities and steel producers. The amounts emitted are more an indication of the amounts produced than of relative efficiency in avoiding emissions. To correct for these flaws in the information the emissions per unit of value added could be calculated. This measure on environmental performance has other perverse effects. Diversification will remove you from the top of any CO_2 list, since emissions of other activities per unit of value added will usually be average. Firms not depending on markets may most easily react to public opinion by putting extra costs on the bill through political or monopolistic mechanisms. Collective waste treatment and electricity production have been organised that way in many countries. Electricity producers do not seem very open to public opinion, however, as indicated by their quest for nuclear power.

Financial instruments: first choice
The *substance deposit* is applicable to the largest carbon flows, see figure 5.5.3, but only for the CO_2 related flows. Implementation is not worked out in detail. It might be very similar to that of cadmium, with the national customs and excise offices as the existing capable organisations mainly responsible for both deposit payments and refunding. By far the largest structural inflow (= inflow from the lithosphere and from chemical forming) is through the extraction of fossil energy resources. The second largest inflow is where calcium carbonates are broken down, freeing the carbon, as in cement production and in many desulfurisation processes. The amount of carbon in carbonates used is a clear measure of potential CO_2 emissions. CO_2 emissions themselves can be measured directly. They cannot be used for the emission tax as these emissions include carbon from fossil sources that has been paid for already, and carbon from the biotic environment, that need not be paid for at all. In all processes currently concerned the decomposition of carbonates coincides with emissions. There then would be no practical difference from an estimated emission tax based on the amounts of carbonates used.

Refunding takes place for stable storage of CO_2 and of CO_2 forming compounds. Waste sites that without further care would retain their carbon content for thousands of years would receive a refund on that carbon. Refunding would take place for all carbon effectively withdrawn from the biological carbon cycle, also for that part that entered the economy from the environment. Underground storage in empty gas fields is technically feasible for CO_2. Pumping CO_2 into the surface waters of the oceans would be as effective in the short term only. This method of storage retains the carbon in the biological carbon cycle however. It only speeds up some transport processes in the environment and is not structural. It is comparable to forestation on formerly barren soils although its time scale is substantially longer. In neither case is the deposit refunded. Forming of CO_2 clathrate structures at the bottom of the oceans, if effectively irreversible, would constitute an outflow to the lithosphere, with a right on refund. The refund could be paid for any compounds formed with carbon, that could be expected to last for several thousand years. Carborundum is a prime example. Plastics in underground sewer pipes might also be a good candidate.

TABLE 5.5.5 THE SUBSTANCE DEPOSIT ON CARBON, CO_2 RELATED FLOWS

deposit payment: ◇ all fossil energy sources extracted ◇ calcium carbonate for iron and cement production and for desulfurisation processes
deposit refunding: ◇ immobilizing CO_2 by well isolated storage, in economy, environment and substrate ◇ immobilizing of CO_2 forming compounds by well isolated storage, in economy, environment and substrate ◇ forming of carbonates ◇ forming of other stable inorganic carbon compounds ◇ forming of stable organic compounds

The deposit scheme is not applicable to methane. For the main flows, the deposit would be levied at production and repaid at combustion. Payment gives no problem. Repayment for decentralised use is administratively hardly possible, however, as measurements there are not very exact. Also, the deposit would be extremely high compared to the value of the widely used methane. Other main flows, forming and emission at anaerobic fermentation of organic materials (in waste sites and in ruminants), cannot be brought under the deposit scheme either.

Emission tax and tradable emission permits

The *emission tax,* based on direct measurement of emissions, may be applied to specific substreams of carbon. First the CO_2 emissions. Large-scale combustion processes burning fossil fuels can be monitored but these flows are covered already by the deposit system. For diffuse emissions measurement is not possible. A special problem is related to emissions originating from extractions from the environment, as biomass. These are not covered by the deposit system. If these emissions are to be paid for, symmetry requires that negative emissions receive a negative tax of the same amount per unit. All agricultural activities would then be subject to the negative tax, while all human consumption and related emissions should be taxed. This is completely impracticable. It also does not make sense if the long-term carbon cycle is to be influenced primarily. If organic flows are excluded, special problems come up where mixed emissions arise which are partly fossil, partly from chemical forming, and partly from biomass. This is the case with large emitters such as cement producers and waste incinerators. The emissions would then have to be distributed over their sources. The realm of emission taxing is then left. The substance deposit here is a preferable alternative, easier in implementation and as good in dynamic efficiency.

For emissions of CH_4 things are different. The option of the substance deposit is not available. Some emissions might be measured more or less directly. Methane in ventilation air from the shafts of coal mines is an example. Perhaps emissions of methane from pipe connections can be measured, by subtracting paid outflow from inflow. This would require better measurement at inflow and outflow of the pipe system, including pipes into households. Consumer organisations have found large measurement errors, often of over ten percent. Precise measurement has become practically possible, however, as new electronic measurement devices have reached the market[1].

Tradable emission permits (and offsets) are applicable to CO_2, according to the OECD. Their reasoning is that emissions may be measured from the products burned (OECD 1991, p.94). Permits for carbon, i.e. CO_2, emissions do not exist now. A permit system for acquiring and using energy resources could be introduced of course, based on their carbon content. After introduction, these permits could then be traded. However, all transactions with energy resources would then have to be registered. The simplest form would be that any transaction requires the sale of a permit for the use of that amount in the time period specified in the permit. Controls on factual emission would still be necessary, as otherwise the non-emitting techniques would not be covered and not be stimulated by the permit system. It would be an extremely tedious and costly operation, if at all possible in peace time[2]. Thus, tradable emission permits are not applicable to the main emissions of CO_2.

Financial instruments: second choice
The *estimated emission tax* is always applicable where the emission tax is. A number of minor flows would be the prime domain of the estimated emission tax as a second choice financial instrument. Relevant for CO_2 emissions are the pre-production losses of oil and coal. The small-scale production of peat, for heating, and lowering water tables in peat areas are examples where the estimated emission tax might be applied. Lowering water tables is a special case in that no carbon is handled. The effect on carbon is indirect only. In that sense it is comparable to acid emissions that indirectly cause the weathering of limestone. Acid emissions require more research in this respect and have been excluded from the climate analysis here. Lowering water tables is given a place here, with a question mark.

The estimated emission tax is the main instrument for methane emissions. Important applications are at the mining of coal and lignite; leaking at the transport of methane, especially at retail distribution; and anaerobic microbial decomposition of organic materials. Landfill and animal husbandry, specifically in ruminants, are main examples of this decomposition. Implementation seems awkward with animal husbandry. Fermentation within ruminants might be brought under the estimated emission tax. Fermentation of manure, is highly variable in the amounts of methane produced and cannot easily be brought under the estimated emission tax. Paddy fields are an important source as well. The effect of making paddy fields is quite indirect, as with CO_2 emissions from lowering water tables in peat areas. Ruminants and paddy fields are included with a question mark.

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[1] Personal communication with Perfors, researcher at the Dutch consumer organization Consumentenbond. Precise electronic measurement would have the added advantage of easy feed back on behaviour to the consumer. This can lead to more energy conscious behaviour. Introduction of these devices might be classified under cultural instruments above.

[2] In wartime such licences have been usual for basic food and energy, mainly for reasons of distributional equality. Such systems are not advocated for permanent use by any environmental group. The reasoning seems contaminated by an analogous application to internationally agreed state quota for emissions. Between states these quota for nationally allowable emissions could be traded. Such a system would not regulate the behaviour of emitters. It would make the internationally agreed obligations of governments more flexible.

Financial instruments: third choice

Financial instruments discussed in politics often are product taxes[1] and subsidies. Two proposals related to carbon have come to the political fore, see Appendix 1. The European Commission proposes a tax on energy resources, one half based on energy content and one half based on carbon content. The Dutch government has commissioned a study on similar taxes, also one half based on energy content, the other half on carbon content. The carbon part in these product taxes on energy cover only one half of the working of the substance deposit, as an indirect emission tax. The carbon part in the taxes induces substitution between different types of fossil energy and towards nuclear energy. The energy and carbon part each give an inducement for substitution towards non-fossil and non-fissile sources of energy and towards energy saving. Since energy depletion is no problem, the taxes could better be aimed completely at the carbon part of fossil fuels. All effects of that energy tax would then be fully covered by the substance deposit on carbon. Compared to the deposit scheme, however, the energy taxes lack the incentives for fossil carbon storage and do not influence the very substantial non-fossil fuel related carbon flows at all. For the time being, the storage option seems more attractive economically than solar energy and wind energy as a means for reducing CO_2 emissions. No one can tell which of these main options are the most attractive in the long run. If the energy tax is chosen, no one will ever know if the options for binding and storing carbon were the more attractive ones, as the relevant techniques will never be developed commercially. Compared to the substance deposit, the energy source taxes, even if fully directed at carbon, thus are inferior to the deposit scheme. Administratively they have another disadvantage. As an input tax the energy taxes proposed effectively tax the property of others, e.g. the property of the oil producing and exporting countries. This might pose problems in terms of international law and in terms of international politics. Emission taxes and the fully equivalent substance deposit system would not or much less exhibit these disadvantages.

Product taxes might take into account the contributions to global warming in the whole life cycle of the product. If so, they can be levied on final products only. If intermediary products were also to be taxed, an unacceptably vague system would result with numerous partial overlaps. Plate steel would be taxed for its CO_2 emissions, as would the parts made of steel and again the car in which the parts are mounted. Part of the car thus would be taxed three times. Setting up a system of compensations would be extremely complicated. It would require a full equivalent of a VAT-like administration. This administration not only would have to cover the national state, as with VAT. It would have to cover all states that contribute to the production of the car, that is the whole world. A product tax could also be restricted to the life span (not to be confused with the life cycle) of the product to be taxed. This option may be internally consistent, taxing any installation for its own emissions upon operation. Then the product tax becomes a simple form of the estimated emission tax. If not, the taxing basis is arbitrary and unequal and thus unjust. As a consequence the effectiveness will be limited, at excessively high costs per unit of emission reduced.

[1] As in most proposals for Ecotaxes, see for example Teufel et al. (1988).

In special cases subsidies might be a reasonable third-choice option. They may be attractive only if emission taxes and equivalent substance deposits do not yet cover the field effectively. These first-choice general financial instruments would make technological development profitable, with a much more impartial and indiscriminate pressure towards technological development than any other instrument. For subsidies, the MITI approach as practised in Japan (van Wolferen 1991) might restrict possible negative effects as much as possible. In that approach all potential producers are brought together. They define the first stages of research towards development that will be a combined effort, with results open to all. Afterwards, in the commercializing phase, each has its own private non-subsidised development[1]. By including all potential producers, the disadvantages of indirect effects of subsidies are minimised. Nevertheless, those that invested in the subsidised field already would have wasted their money. Very generally, subsidies have positive effects on the objects directly applied to, but negative effects through more indirect mechanisms that may or may not reverse the direct effect. All research contributing to a collective good, here lessened risk of global warming, would become riskier than if the subsidy instrument did not exist. Broadly occurring negative indirect effects with small effects per decision influenced, are extremely difficult to model. Their broad working could easily offset any concrete positive effects.

Direct regulations of general applicability

General rules on energy efficiency of products are prepared by the European Commission. If sanctions seem realistic this might improve the energy efficiency of these products effectively. Tightening emission standards in California in the Seventies and Eighties are an example of how effective such a policy might be. With energy efficiency the results per apparatus might be similar, usually requiring some extra costs. These extra costs partly translate into extra energy requirements. As energy saving saves income as well, there may also be a net income gain. This will be spent on more of the same or on other items. With cars, for example, the ultimately resulting net reduction in energy use might be small. First, better fuel efficiency may be realised by replacing steel with aluminium, at higher energy costs in production[2]. Secondly, with lower energy costs per car, people will buy more cars that together might use more fuel and might spend more on other items that use energy like luxury food and long distance holidays.

However, direct regulations may play an important role in preparing systems changes. Electric cars will not become developed seriously, even if solar energy could take care of all electricity production at prices somewhat lower than current production costs. A large scale systems change is then required. Even if the cars were developed, with the battery problems solved, introduction could be postponed indefinitely in oligopolistic (and also in

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[1] US battery research, triggered by the Californian law requiring emission free cars, seems to follow this Japanese pattern. Car manufacturers formed the Advanced Battery Consortium, for collectively developing new types of batteries. The consortium is financed by the US department of Energy for 50%. Contracts for development work go only to American companies. It is not so clear however, how the partners in the consortium will use the results. They might treat it as collectively owned, with proceeds of the technologies developed shared according to the share in the consortium. Then effectively a probably temporary monopoly has been created. In Japan, the final development work is done by all participants separately, with competition as to who is to receive the fruits of the research done collectively. See the report by Charles 1992.

[2] Currently, the energy requirements for manufacturing a car are around ten percent of the energy required for driving during its entire life span, according to Mol (1993).

more competitive) markets. No single car producer would pay for the costs of setting up recharging facilities for a small proportion of only his car production. Without these facilities, no serious customer would buy an electric car as a substitute for current gas and petrol vehicles. Regulations as in Californian law may speed up such a system development. Knowing that the market will be there, the recharging facilities will be installed by individual firms, by oil companies, or by the car sellers, possibly together. Such regulations are most effective at solving bottlenecks in the introduction of new technologies. The substance deposit remains the driving force behind the change.

Of course, regulations may be effective by themselves. However, the all-pervasive nature of carbon, both through its versatility in chemical forms and applications and through its extremely general applicability in the form of energy, makes it very difficult to get a hold on total emissions. Improvements in energy efficiency by direct regulations on products and individual processes are plagued by the problems mentioned above. The inelastic supply of fossil resources, substitutions in supply of specific commodities and demand shifts, together, may lead to similar emissions from other activities as before policies were implemented. Only the prescription of end-of-pipe techniques may lead to reasonably certain emission reductions. Utilities could be forced to store a certain percentage of the CO_2 they produce[1]. Waste handling could be allowed only in forms that store the carbon more or less indefinitely. Such policies choose one specific option. That option might, or might not be the most efficient, now or in the future. Financial instruments leave the choice between energy resource substitution, increased energy efficiency, diminished end use, and end-of-pipe solutions to the adjusted market and to the dynamic development of technologies in it.

Direct regulations with limited applicability

The climate problem is a purely global one. There thus cannot be environmental reasons for differentiating between different emitters at different locations. Carbon emissions may create problems at the local level where they are emitted. These are not climate problems but problems related to for example toxicity or ozone forming characteristics of the hydrocarbons emitted. Global warming emissions do not have direct local effects of course.

5.5.7 The flexible response strategy: financial instruments

In developing the flexible response strategy, first the instruments are chosen. Then, for financial instruments, a reasonable level of the tax is defined, directed at a substantial reduction of global emissions in the next decades and thereafter. Finally, here is an estimate of the potential effectiveness of the *flexible response strategy*, based on a number of technical assumptions.

Instruments

Liability can hardly play a role in reducing the climate problem. Some changes in rules on landed property might be very effective in a number of Third World countries, preventing deforestation and erosion. Cultural instruments may be useful in circumstances

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[1] Other forms could be the obligatory storage of a certain amount of CO_2 per kWh produced or a limit on total amounts of CO_2 emitted per kWh produced.

where the environmental impact of a product in its whole life cycle are restricted to climate effects only. Thus the role of this type of instrument is very limited as well. Financial instruments are broadly applicable. The CO_2 related flows of carbon into the economy from the lithosphere and back to the lithosphere can be brought under the working of the *substance deposit* nearly completely. Accumulation in inert stocks in the economy may brought under the deposit system as well. Only some smaller streams are difficult to measure. Small-scale peat production and consumption, e.g., cannot easily fit into the deposit scheme. It might be brought under the *estimated emission tax*, or left out of the system altogether. Emissions of oil at exploration and at exploitation, before it reaches the pipes in which it is measured, can only be estimated. The *estimated emission tax* is the instrument applicable to such pre-production losses.

The inflow from chemical forming, amounting to 2.5% of total carbon inflow, can be covered by financial instruments as well. The inflow for cement and plaster production may be brought under the deposit system, as may be carbonates emitting CO_2 when used for binding sulfur. Oxidation of peat by lowering water tables can be quantified only indirectly, on the basis of highly differentiated local data[1]. For these flows again the estimated emission tax is applicable. The carbon inflow from immobile stocks is thus covered for well over 99%.

If carbon wastes are stored as immobile stocks in environment or substrate, repayment of the deposit is due without many administrative problems. Criteria for assessing the long-term stability of waste storage are needed. Some carbon compounds, such as carborundum, are so stable that creating them constitutes an outflow, even if they remain in the economy. They could receive a refund as well, without much complexity of implementation if criteria are available.

Methane requires a different mix of instruments. The substance deposit is not applicable. The emission tax and the estimated emission tax are, for different flows. Methane emissions from coal mining differ widely between types of coal, between mines and between different layers within one mine. For closed pits, continuous measurement techniques can provide actual emission data. There the *emission tax* is applicable. For open pit mining direct measurement is not possible. Then the *estimated emission tax* is applicable. On distribution, losses of CH_4 may be measured, indirectly only, or can be estimated. e.g., based on real measurement at an a-select sample of leaks in pipe joints. Methane emissions from animal husbandry and paddy fields make a significant contribution of over fifty percent of the CH_4 related contributions to global warming. With animal husbandry, application of the estimated emission tax is feasible in most industrialised countries. Paddy rice is mostly grown in non-industrialised countries. The implementation of even the estimated emission tax seems quite burdensome there. Further economic development, with its rising labour costs, may lead to the abandonment of the highly labour intensive paddy rice growing. See table 5.5.6 for a survey of the applicability of instruments.

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[1] In the Netherlands the highly capable "Water Boards" might be the one to tax as they fully control the lowering of all water tables in the areas under their jurisdiction. Internationally, this probably is an exception.

TABLE 5.5.6 THE FLEXIBLE RESPONSE STRATEGY FOR EMISSIONS OF CO_2 AND CH_4

Inflow into the economy from lithosphere	Instruments applicable:	Outflow from the economy to lithosphere	Instruments applicable:
-fossil energy extraction	deposit payment	-disposal of CO_2 in gas domes and clathrates	refunding of substance deposit
-pre-production losses of oil and coal	(estimated) emission tax[1]	-immobilised disposal in substrate or environment of CO_2 forming compounds, esp. wastes	refunding of substance deposit
-small scale production of peat	estimated emission tax?		
-breakdown of organic materials into CO_2 by lowering water tables in peat areas	estimated emission tax?		
Chemical forming in the economy[1]	Instruments applicable:	Chemical destruction in the economy	Instruments applicable:
-decomposing calcium carbonates at cement, iron and steel production	estimated emission tax	-stable inorganic compounds (carborundum, etc.)	refunding of substance deposit
-binding sulfur with lime in energy production	estimated emission tax	-stable organic compounds	refunding of substance deposit
Inflow to the economy from the environment	Instruments applicable: **not to be regulated**	Outflow from the economy to the environment, CH_4 only[1]	Instruments applicable:
		-microbial breakdown of organic materials into CH_4:	
		*landfill, oil	estimated emission tax
		*animal husbandry	estimated emission tax?
		*paddy fields	estimated emission tax?
		-pre-production losses of methane production	estimated emission tax
		-methane from coal and lignite mining	(estimated) emission tax
		-distribution losses of methane	(estimated) emission tax

How do the different instruments cover the current flows? The substance deposit has a prime role in the strategy for all CO_2 related flows. It covers over 99% of all structural carbon contributions to global warming. For CH_4, with a current share in carbon related global warming contribution of around fifty percent, the emission tax and the estimated emission tax may cover the main flows.

[1] Where extraction or forming coincide with emissions, only one of these is given in this survey of the flexible response strategy.

Together, these three instruments cover virtually all contributions to the CO_2 and CH_4 part of the climate problem. They stimulate virtually all solutions possible in an equal manner.

Deposit / tax level

The combined effectiveness of the flexible response strategy depends on the level at which the tax is set, equal for substance deposit, emission tax, and estimated emission tax, per unit of GWP involved. For methane, with a global warming potential of 21 it thus is 21 times as high as for CO_2[1]. Suppose that the equivalent to the planned EC tax of $10 per barrel of oil equivalent, in the year 2000, were levied solely on carbon emissions, through the substance deposit. Taking oil as a reference, this amounts to ECU 0.067 per kilogram of carbon[2]. First computations of the costs of storing CO_2 underground amounted to ECU 0.073 per kilogram of carbon stored. At the $10 equivalent for the carbon deposit, underground storage is just not profitable yet. A carbon deposit equivalent to a deposit on oil of $10.95. the is required. Induced changes towards solar energy will require time. Profound system changes imply large investments that will take place only after deliberation and experiment. Manne and Richels compute a long-term level of ECU 0.39 per kilogramme of carbon as the price for a substantial change-over to solar energy, as the combination of market price plus energy tax. The estimates given here are much more optimistic. They are, first, based on a conservative extrapolation of the downward trend in the cost price of solar cells. Secondly, there is the assumption that costs of storage and transport will not be much higher than those of current energy systems, *after* the systems change has been made for a substantial amount of energy. It then is a matter of time, say a decade after introduction of the taxes, until large scale introduction of solar energy will occur. Dropping prices of fossil energy will slow down the further introduction of solar energy in new domains of application.

At the deposit/tax level of ECU 0.073 per kg C, CO_2 storage becomes an attractive option. The minimum amount of *the tax thus is to be put at ECU 0.073 per kg carbon*, to cover the costs of this first main technique for structural outflow. This seems the most reasonable tax level to arrive at if really substantial reductions in emissions of climate gases are desired. The tax would amount to around $11 per barrel of oil, with a full repayment if the CO_2 formed were fully stored. This level would also cover some extra costs of storage and transport of solar energy. Deposit payment on, e.g., oil would be slightly higher than the energy tax now proposed by the European Commission at $10, but lower than the one that has been discussed in the US Senate, $13. The tax studied by the Wolfson Commission, A variant, amounted to $40 per barrel of oil.

Effects of the flexible response strategy on global warming

To make the change to a substantially lower level of carbon emissions, many vested interest will become harmed and will try to postpone effective introduction. Inflexibility will also arise because of existing regulations (see Tellegen 1989). One example are the problems faced by many wind power producers to sell their electricity to the national

[1] In the GWP of methane the effect of CO_2 eventually formed from it is included. If the methane originates from fossil stocks, that part has been paid for already, as a deposit. For these methane emissions a factor of only 20 then should be used. See Lelieveld and Crutzen 1991 for a similar position on the GWP of CH_4.

[2] 0.073 ECU/kg C x 1.219 $/ECU x 0.86 kg C/kg oil x 0.9 litre of oil/kg oil x 159 litre oil/barrel oil = $10.95

grids. Opening up trade in energy, using the existing electricity grids, also for smaller quantities, is already on the agenda of the EC. Also, 'sunk costs', in capital goods, will delay changes till new investments are due. If the deposit/tax system were introduced in ten years time, with instalments of ten percent per year, the full application would only start after ten years. Technology development, however, would start tomorrow. Technology development would go on indefinitely, exerting a tendency towards lower emissions as long as the system functions. Autonomous rise in emission levels, mainly through a rise in consumption, would detract from this trend and could reverse it eventually. Predictions of effects can be made only very provisionally. The most sensible approach, at least with the limited means available here, is to make some technical assumptions, based on the tax level of ECU 0.073 per kg carbon. The assumptions create a possible world say 30 years from start of the scheme. Political discussions before introduction take another decade. Thus, the year 2030 is depicted.

The quantitative assumptions for that year are the following:
1. Final energy consumption (excluding wood) rises with two percent per year, rising to double the 1990 amount, that is to 800 EJ.
2. All coal-fired power stations are of the type now built as combined cycle coal gasification plants, with zero emissions of CO_2 (and other contaminants). Ultra heavy oils and other dirty fossil resources are used only in these or similar installations[1].
3. Fifty percent of all energy consumption is based on solar power.
4. Twenty five percent of energy consumption is based on fossil methane and some light fuels, without CO_2 storage[2].
5. Twenty percent of energy consumption is based on dirty fuels burned cleanly, as described in '2'.
6. Five percent of energy consumption is based on nuclear power, assuming a modest growth in nuclear power production of two percent per year.
7. Consumption of calcium carbonate for cement and iron production doubles. It stops for the binding of sulfur. This will not be required any longer.
8. Fuel wood will hardly be used any longer since in tropical countries solar based electricity will take over.
9. All emissions of fossil methane will decrease substantially, e.g. by cleaning ventilation air in coal mining and by shifts towards other mines.
10. Most emissions of biotic methane will decrease. Only ruminants will remain at their substantial emission level.

As a result of these shifts, carbon extracted from fossil resources will still rise, although much less than if current trends continue. The production of fuel wood will decline steeply. For all other purposes, wood production will go up somewhat. Food production covers the requirements of a rising population. A main question is what will happen to manure. It is supposed here that part of the manure will be treated as a waste. After processing for the recovery of valuable components, the remaining carbon part will be stored in an immobilised form, as a waste.

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[1] Additional environmental advantages are that all organic chemical wastes now burned separately, stored or emitted can be co-burned at hardly any extra costs. Virgin sulfur production becomes superfluous.

[2] Since the combustion engine will also become obsolete, cracking of oil will not be necessary any more. Refineries could then become much simpler installations.

In this new situation the carbon related structural contribution to global warming drops by fifty-six percent as compared to the 1987 level, see table 5.5.7 for the emissions and global warming contributions resulting. The difference with the autonomous development will be much larger of course. A drop on the order of seventy-five percent could then be registered towards 2030.

TABLE 5.5.7 REDUCTION IN GWP CONTRIBUTION OF CO_2 AND CH_4 THROUGH THE FLEXIBLE RESPONSE STRATEGY, ROUGH ESTIMATE OF EMISSIONS

Outflow from the economy to the environment		related to: CO_2	CH_4	abs.	GWP-contr perc.[1]
-Combustion processes					
*fossil fuels		4400	?	4400	39.1%
*wood		97	3	160	7.1%
*wastes	plastics	0	?	0	0.0%
	organic	0	?	0	0.0%
-Biotic processes					
*human respiration		0.04		0	0.0%
*animal respiration		<1		1	0.0%
*from ruminants directly			60	1260	%
*rice growing in paddy fields			20	420	%
*organic materials (landfill)			1	21	4.6%
*net oxidation of peat		15		15	0.2%
-Chemical forming *cement		250		250	0.8%
*plaster		49		49	0.2%
*industrial. applic.		1		1	0.0%
*oxidation of carbon in steel		0		0	0.0%
-Organic materials to environment					
*biomass, sewer		10	?	10	0.3%
manure		100	?	100	4.0%
*pre-production losses					
oil		5		5	0.2%
coal		10		10	0.2%
*fossil methane spills			35	735	16.6%
-Net oxidation of agric. soils					
*erosion		100		100	1.7%
-Deforestation		500		500	10.5%
Subtotal	*CO_2*	*5538*		*5538*	*61.6%*
	CH_4		*119*	*2499*	*38.4%*
	Total carbon	*5657*		*8037*	*100.0%*
Net emissions	**CO_2**	**2478**		**2478**	**50%**[2]
	CH_4		**119**	**2499**	**50%**
	Total carbon	**2597**		**4977**	**100%**
Total reduction in GWP_{100} as compared to 1987 (see table 5.5.4)				**6220**	**56%**

[1] As a percentage of total GWP emitted.

[2] As a percentage of net emissions.

5.5.8 Conclusions

Results of the analysis

There is no energy depletion threatening even for the very distant future. Fossil resources and nuclear energy could, each independently, supply our current consumption for thousands of years. Solar energy, probably becoming competitive within decades, can take over a substantial part of energy production in the decades then following, if prices of fossil energy do not fall too much. Solar energy, in the long term, can easily cover a multiple of current energy use, indefinitely.

The climate problem is a serious problem because the chances of substantial climate changes now are well above zero. Some short-term changes and sheer luck may allow us to get through the next decades without major calamities. For the next decades the carbon based human effects on global warming are primarily dependent upon the emissions of methane, not of CO_2 related emissions. In the long run, these two types of contribution are more equal.

The quantitative analysis here has been directed first at the long-term prevention of CO_2 emissions. In that analysis, the carbon flows circulating between economy and environment are not so relevant, only the long-term effects count. These are determined by the structural inflow, from the lithosphere and from chemical forming the economy, to be reduced, and by structural outflow, by the stable storage ultimately in the lithosphere and by chemical destruction, to be increased. The emission reductions of CH_4 may be realised mainly in a more conventional manner, through closing leaks and burning methane from remaining ones.

The analysis of technical solutions has not led to many positive results. The main message is that the mode of analysis is decisive for the results, with more complex modelling indicating the minor effects. All micro effects are dampened at the macro level. Measures "in the middle of the system" are most prone to unclear results, as is the case with most recycling in the context of CO_2 reductions.

Energy taxes now widely discussed should not be based on energy content, since there is no depletion problem at all, but on carbon content. They then still are a second choice option only. Bringing carbonates under the tax and also paying for any structural outflow of carbon would transform the carbon based energy tax into the substance deposit on CO_2. Some problems of introducing the taxing system as encountered now in international fora could then be avoided. The environmental performance would improve drastically by these additions.

The widely accepted Hotelling rule for pricing of depleting scarce resources would only indicate an infinitely small price rise for fossil energy resources, in the order of magnitude of 0.1% a year. If energy taxes are applied to carbon at a much higher level, as discussed, they are not the most suitable instrument for large emission reductions, as they do not stimulate immobilization and chemical destruction of CO_2 related carbon flows. They cannot reduce the more important methane emissions at all.

The flexible response strategy first works through case independent structural and cultural changes. Changes in the rules on landed property in most Third World countries are a main structural instrument for short-term reductions in the burning of biomass existing in the environment, in natural forests. Liability cannot play any role in the carbon related climate problem. The case specific instruments can be restricted to financial instruments again. Long-term emission reductions of CO_2 can be induced by the substance deposit, through reduced structural inflow and increased structural outflow, with a possible, but not strictly necessary, minor role for the estimated emission tax. Reductions of methane emissions cannot be realised with the substance deposit. A main part of these emissions can be covered quite well with the emission tax and the estimated emission tax, with ruminant and paddy rice growing as the most difficult objects for even the estimated emission tax.

At a level of around ECU 0.075 per kilogramme of carbon, the halving of current GWP contributions in forty years is a reasonable first estimate of effects, based on some main technology changes. There are no special administrative problems involved in implementing these financial instruments.

APPENDIX 5.5.1
ENVIRONMENTAL TAXES ON ENERGY, OPTIONS

Energy taxes proposed
The Commission of the European Community has proposed an energy tax. It is partly based on the energy content of the energy carriers, for about a third of its value, and partly based on the carbon content, for the remaining two-thirds[1]. Some large energy users competing with third country producers would be exempted from the tax. The amount discussed is equivalent to $3 per barrel in 1993, increasing for several years by one dollar per year, reaching a tax level of $10 per barrel in the year 2000[2].

In the Netherlands, there are similar proposals. A governmental commission, the Wolfson Commission, has, aided by the Central Planning Bureau, investigated several energy taxing schemes. The first, variant A, is an OECD-wide tax on energy carriers. It would double the Dutch energy expenses. The taxing schemes apply 50% to energy content and 50% to carbon, see table 5.3.1 for the resulting taxes on several primary energy sources and secondary energy carriers. The tax is also levied on imported secondary energy carriers. A similar tax, variant B, is intended to be introduced throughout the Netherlands only. Both schemes would lead to substantial reductions in national income, see table 5.3.2, because the model used, predicts that industries will move to places where these energy taxes do not exist. A third scheme, variant C, also at the Dutch national level, avoids the negative side-effects of the other two as much as possible. It taxes only end users and these only to what for most firms is a small amount of consumption. Effectively, households pay about fifty percent of the total tax amount and mainly smaller firms, in locally restricted markets, pay the rest. For larger industries this tax is insignificant. All economic activities that still might be hurt by international competition receive an exemption. Examples are greenhouses, which together use much natural gas in the Netherlands. Diverging from the EC taxing scheme, that used an introductory period of over seven years, all three schemes, A, B and C, have been introduced in one hypothetical move. Variants A and B hardly contain any mitigating or compensating measures for those industries particularly hurt, another aspect that diverges from the EC proposals. Variant C permits all mitigating and compensating changes in the purely Dutch tax on final energy use. The Wolfson Commission did not investigate any taxing schemes at the administrative level of the EC. The three taxing schemes have also been quantified for a lower tax rate of fifty percent.

How do the levels of the taxes investigated by the Wolfson Commission compare with other proposals? They are by far the highest in the discussion. The German proposal by Teufel et al. comes second, at around a third of the Wolfson level. The American proposal discussed by a the Congressional Budget Office is slightly lower, with a carbon tax amounting to $13 per barrel of oil. The European Commission comes fourth at around a quarter of the level of the Dutch schemes. The English-Norwegian proposal by Kverndokk is slightly lower. Their long term switch prices are substantial lower still.

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[1] The first public action by the European Commission on this subject was a Working Paper presented to the Council of Environment Minsters of 21/XII/90. One political reason for the mixed nature is that without a tax on energy content nuclear energy would become very profitable. The final proposal is that of 14 October 1991 (Commission 1991). In it a proportion of energy taxes to CO_2 taxes of 1:3 has also been mentioned.

[2] On 13 May 1992 the Commission decided to proceed with the tax on the condition that the US and Japan would introduce similar taxes. The level of the tax would rise to $10 per barrel by the year 2000. The base of the tax was changed from one-third to one-half the energy content.

TABLE A.1.1 PRICES AND TAX LEVELS FOR SEVERAL PRIMARY AND SECONDARY ENERGY SOURCES IN ECU PER GIGAJOULE ($=10^9$ J), STUDIED BY THE DUTCH WOLFSON COMMISSION

ENERGY CARRIER	CURRENT DUTCH PRICE, INCL. CURRENT EXCISES	TAX A/B, 100% LEVEL	INITIAL PRICE INCREASE BY TAX A/B	TAX C, 100% ONLY SMALL USERS	INITIAL PRICE INCREASE BY TAX C
coal	1.71	7.90	461	--	--
crude oil	2.86	6.84	240	--	--
heavy fuel oil	2.71	6.95	256	--	--
diesel oil	10.88	6.84	63	6.46[3]	59
light fuel oil	6.25	6.84	109	6.46	103
LPG	9.37	6.50	69	6.46	69
petrol	19.33	6.84	35	6.46	33
other oil prod.	9.05	6.84	76	6.46	71
natural gas SU[1]	5.53	6.00	108	4.49[4]	81
natural gas LU[2]	2.75	6.00	218	--	--
other (=uran.)	5.39	3.18	59	--	--
electricity SU	21.04	15.84	75	21.79[5]	104
electricity LU	12.36	15.78	128	--	--

[1]SU = small-scale users
[2]LU = large-scale users

[3]Assumed: 36 MJ/l
[4]39 MJ/m^3
[5]3.6 MJ/kWh

Source: Adapted from Wolfson Commission, Tables 3.1 and 3.3

TABLE A.1.2 COMPARISON OF TAX LEVELS IN DIFFERENT PROPOSALS

	GJ coal	GJ oil	GJ gas	kg C
Kverndokk[1] short term	1.25			1.00
Kverndokk[1] long term (100yrs)	0.48			0.39
European Commission (yr2000)		1.35		
Congressional Budget Office		2.13		
Teufel et al.	2.30	2.33	1.60	
Wolfson (100%)	7.90	6.84	6.00	

[1] Computation based on Manne and Richels (1991), for OECD and (former) Communist Countries together. Coal taxes computed from carbon taxes suppose 80% carbon in coal.
ECU 1.00 = $1.29 = Dfl 2.32 used for converting currencies.

For each of these schemes studied by the Wolfson Commission, a dynamic, mainly macro-economic model, by the Dutch CPB (Central Planning Office) has predicted the effects of the taxes on prices, income, employment, and the use of energy. The net reductions in CO_2 emissions are about proportional to net reduced energy use, so the study assumes. The results are supplementary to the autonomous development path predicted by the models under the assumptions used. As in all modelling, these assumptions may be questioned, as may the model structure itself. An example of a dubious assumption is a price increase of three percent per year, while empirically, oil prices have tended to fall over the last three quarters of a century. However, it seems improbable that another autonomous time path modelled would have a severe influence on the predicted effects of energy taxes. There is one specific omission in the model structure that has severe consequences for the value of the outcomes. There is no reaction in the models in terms of a change of technology other than general efficiency increases and shifts between types of fuels used. Thus, results do not reflect currently available but not generally profitable technologies nor future developments in technologies. As an example, most sources say that solar power cannot yet play a significant role in the next two decades (see for example Langeweg 1989 and Odell 1990). For the more distant future such assumptions do not hold, especially not if, e.g., R & D outlays change from fusion research to solar power development. This change is currently taking place in the European Community, for example in the Joule II programme. The switch over prices mentioned in the proposals above, especially those by Kverndokk, show that this is a serious omission.

At the global level total effects are quantitatively relevant for the environmental problems of energy use only if many countries join in emission reductions. However, the combination of a high tax rate and broad application, as in tax A at the level of the OECD, also has a severe effect on energy producers. Their proceeds drop for two reasons. Reduced production *and* a lower price is what they face. The tax would cut off the main part of the 'rent' producers currently receive from oil production. The full effect would result after some years of adjustment of production and consumption. Using predictions by CPB (1992) for the Wolfson Commission it can be shown that by 2015 all oil producers, including the OPEC countries, would see their incomes from oil fall by sixty-three percent compared to what they might have expected to result from autonomous development (the "reference path" in CPB terms)[1]! For coal producers the reduction in proceeds would be even more extreme, for producers of gas, less extreme.

How to evaluate the policy schemes

When evaluating policy measures, their effectiveness, efficiency and equality should be assessed. As to effectiveness, the first question to answer is which effects are to be assessed. The stated aim of the taxes is to reduce energy depletion and to reduce CO_2 emissions. If we disregard depletion for the moment, effectiveness is only in terms of reduced CO_2 emissions. It is not in terms of reduced global warming potential. Any problem shifting, especially through the increased emissions of methane, thus remains invisible. The next question is the scope of CO_2 emissions that has to be taken into account. To allow comparisons between the three policy variants it is the effectiveness of Dutch policy implementation that should be assessed. In that case, however, Dutch emission reductions are not the relevant item. These reductions are partly the result of transferring Dutch industry abroad. This does not reduce global emissions at all. Thus, effectiveness is the net global emission reductions caused by Dutch policy implementation.

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[1] This effect is based on a 60% decline in gross sales prices for producers (CPB 1992, figure IV.4) and a 13 % increase in the share of oil in global energy consumption, see CPB 1992, figure IV.3, and a reduction of 22 % in total volume of energy consumption, table IV.11, all figures as compared to the reference path of CPB.

Efficiency of a policy measure, its cost effectiveness, is the ratio between costs and effectiveness. Again, to allow comparisons, the cost effectiveness of the Dutch part of policy is the relevant item, since only Dutch costs are to be computed. Costs may be defined in a number of ways. The Wolfson Commission has chosen an aggregate effect on national income (and similar measures) in constant prices, computed through a macro-economic sectoral model. A global energy demand and supply model gives the input of energy prices into this model of the Dutch economy. Before the model can run its course into the future, some macro-economic policy has to be specified. In the policy schemes, all proceeds from energy taxes go to the lowering of taxes (and social security levies) on personal income. Simultaneously, the assumption is that energy taxes do not influence the budget deficit. To create this result, other taxes had to be adjusted. Thus, the macro-economic effect of energy taxes is not only based on the policy measure, i.e. the energy tax itself but, also on additional assumptions on macro-economic policies and specific taxing policies. Another factor that determines the outcome of environmental policy is the level of unemployment when introducing that policy. Thus, environmental policy, in the models, is part of macro-economic policy, contrary to the usual analysis of emission reductions in micro-economic terms. The nature of predictions of macro-economic effects of policy measures that usually would be seen as micro-economic measures is thus somewhat vague. These effects are the combined result of implementation of the environmental policy instrument on the one hand, and of macro-economic policy and macro-economic starting conditions on the other. Nevertheless, it may be interesting to analyze the outcomes of the three variants as predicted macro-economically.

The specification of equality, as an equal environment-economy trade-off in all decisions affected, depends on the specification of the problem regulated. If the problem is global warming, and not, as incorrectly assumed, CO_2 emissions from fossil energy, then that CO_2 does not even cause half the problem. It seems unjust to single out one sector and one group of activities that is responsible for less than half the problem. The carbon part of the taxes is therefore not just. If the problem is energy depletion, the energy part of the taxes is quite just. Since global warming is the problem, all substances contributing to it should be treated equally, in terms of costs induced per unit of environmental improvement. The compensating lowering of other taxes may be disregarded when assessing the justness of financial instruments.

Results of the Wolfson Commission

The outcomes of the models applied by the Wolfson Commission, in terms of effectiveness and efficiency, are quite shocking. It appears that due to the open nature of the economy of the Netherlands and also the OECD, the costs of variants A and B in terms of gross national product lost are astronomic. Variant A causes a reduction in national product in the Netherlands of seven percent, equal to the induced global emission reduction of about seven percent[1], in the year 2015. That is a loss of thirty billion ECU in that year, after twenty five years of adjustment[2]. How can a mere shift in taxing basis, from labour to energy, have such a severe effect? Let us first look at the micro-economic behavioral changes set in motion by the taxes. A kilogram of carbon-in-oil costs ECU 0.18. In the variants A and B this amount of oil has an energy tax on it of ECU 0.38[3]. Through volume adjustments and changes in the composition of energy resources used, a reduction in the emissions of CO_2 results. At a micro-economic level, emission reduction costs may be expected to remain below the tax level. Now, at the macro-economic level, the predicted costs of reduction of CO_2 emissions amount to ECU 10 per kilogram of carbon! An emission

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[1] As a percentage of predicted Dutch energy consumption in the reference path.

[2] The model assumes the policies to be introduced in 1990.

[3] At current oil prices of $18 per barrel; $1.219 per ECU; 159 litres per barrel; 0.9kg oil per litre oil ; and 0.86 kg C per kg oil, a kilogram of carbon-in-oil costs ECU 0.12. This corresponds to an amount of 1.28 kg of oil, equivalent to 55 MegaJoule. The tax amount paid on this 55 megaJoule of oil is ECU 0.38, see table A.1.1.

reduction of one kilogram of carbon could result for example from not using the equivalent amount of oil to evade the tax. This would lead to an income reduction of more than *twenty five* times the tax amount evaded. This result obtains while simultaneously assuming that many energy-saving activities can pay for themselves completely or almost completely, thus creating possibilities to evade the tax with low levels of costs, if any. If mere shifts in taxation would have such prohibitive effects, what then would be the consequences in the models used for raising taxes? How could such a high taxing country as Sweden be one of the richest in the world? The reasonableness of using such macro-economic outcomes as a measure of long-term costs of policy may seriously be doubted. It might be interesting to see what the models would predict if the EC doubled the price of milk and grain, or if the Dutch government doubled its investment in rail infrastructure.

Variant B, the same energy taxes as in variant A but now implemented only in the Netherlands, gives a much more favourable result. Contrary to expectations, income losses after twenty five years, at twenty-six billion ECU, are about fifteen percent lower, while, also against expectations, the emission reductions are nearly twice as high as in variant A. The costs in lost national product are a bit under five ECU per kilogram C not emitted as CO_2, still over ten times the tax paid on that same amount of carbon in fuel. The Central Planning Bureau (p.126) explains the low costs after twenty- five years by an accelerated adjustment process due to high costs in the period just after the introduction of the variant B taxes. A second factor is the possibility to accommodate the rise in energy prices by lowering wages, a possibility not effective in variant A. There, all foreign OECD competitors were in the same situation.

Variant C makes a still larger jump in efficiency. The costs of this tax on final use of energy in the Netherlands, only paid by those that cannot substitute by foreign production and consumption, are comparatively negligible, a bit over four hundred million ECU in 2015. The effectiveness in terms of reduced CO_2 emissions is a bit lower. With five percent it is still in the same range as the other two variants, see table A.1.3. The efficiency rises, equivalent to ECU 0.19 per kilogramme of emission reduction.

The Wolfson Commission did not assess the equality of the taxes proposed. Distributional justness is impaired by any partiality in the application of instruments. Singling out Dutch consumers and small firms as those who have to pay, as in variant C, leads to international inequality. Variant B is better in that respect, but singles out Dutch producers and consumers as against those abroad. Variant C, the OECD-wide introduction is better. All schemes single out part of the global warming problem and are more unequal in that respect.

Discussion
Which lessons can be learned from the energy taxes studied by the Wolfson Commission and the CPB?
First, that macro-economic models predicting long-term effects seem too unstable to reach reliable effects, both in terms of the order of magnitude computed and comparatively between variants. It seems utterly improbable that after 25 years of micro-economic adjustments the costs of the taxing shift would still be so astronomically high. A parable may further suggest why the use of macro models for predicting long-term costs of financial regulation may never be appropriate. If a ship hits a large wave it will deviate from its course. A model then may predict that it will never reach its destined harbour, given all the actions the captain and his crew would have executed without the wave, and some limited corrections on these. However, ships, in the end, usually arrive at their chosen destination, even if they encounter large waves. The macro-models used by the CPB try to predict the effects of waves as cleanly as possible. For that reason they refrain from simulating the compensating governmental reactions as much as possible. Otherwise the effects of

TABLE A.1.3 EFFECTS OF TAXING SCHEMES A, B AND C (100%) ON GNP, DUTCH AND GLOBAL ENERGY CONSUMPTION, GLOBAL CO_2 EMISSIONS AND COSTS, IN GNP PER KG C EMISSION REDUCTION, IN THE YEAR 2015

ENERGY TAXING SCHEMES, 100%	REDUCED DUTCH NATIONAL INCOME[1]	REDUCED DUTCH ENERGY CONSUMPTION	INDUCED REDUCTION IN GLOBAL ENERGY CONSUMPTION[2]	INDUCED GLOBAL EMISSION REDUCTION[3]	COSTS OF EMISSION REDUCTION[4]
Variant A, OECD	7 % ECU 30.6 . 10^9	35 %	5 à 10 %	7 % 3 . 10^9kg C	ECU 9.91 per kg C
Variant B, the Netherlands	6 % ECU 26.3 . 10^9	50 %	10 à 15 %	12 % 5 . 10^9kg C	ECU 4.74 per kg C
Variant C, the Netherlands	0.2 % ECU 0.43 . 10^9	5 %	5 %	5 % 2.2 . 10^9kg C	ECU 0.19 per kg C

[1] Also based on CBS 1992
[2] As a percentage of predicted Dutch energy consumption
[3] As a percentage of predicted Dutch CO_2 emissions, based on index of European energy consumption CPB 1992, table II.5
[4] Dutch costs in terms of GNP lost per unit of Dutch contribution to global emission reduction

General sources: Wolfson Commission 1992, chapters 6 and 7; CPB 1992, chapters II, IV, V and VI

environmental policy changes, both at a micro and a macro level, would completely intermix with the effects of the tax. The result of the model computations shows the effect of the tax without sensible compensating policies. As a method of assessing long-term effects of environmental taxes this does do not seem an appropriate approach. However, mixing sensible macro-economic policy measures with the energy tax dilutes the effects of the tax with all the other policy measures assumed. Results then no longer indicate the effects of the energy tax but only of the mix of several independent policies. I do not think there is a solution to this dilemma in using macro-economic models. At least for the time being the partial analysis of some micro-economic mechanisms, such as costs of emission reduction, are more appropriate for assessing long-term costs than macro-models. Macro-models may play a role in the timing of the introduction of taxes and in suggesting short-term compensating policies.

The second lesson is that there are easy possibilities for using financial instruments that have low efficiency because of not being regulative in terms of the environmental problem to be solved. Substantially depressing volumes produced and consumed as a sole means towards emission reduction will always be much more costly than also reducing emissions by technological means. If the global warming problem is to be prevented, emissions taxes on all global warming substances emitted should preferably be introduced, not just taxes on carbon in energy resources extracted from the lithosphere. Then the as yet unknown costs of new techniques will decide if volume reductions will primarily contribute to emission reductions or if outflow-increasing techniques, such as storing CO_2 in exhausted gas fields, are more attractive.

Thirdly, the conclusion from the former two points is that the picture sketched of regulative taxes is extremely over-pessimistic. Sub-optimum taxes with very limited effects on emissions, combined with exorbitant costs will certainly reveal the "unattractiveness" of environmental taxes.

APPENDIX 5.5.2
LONG-TERM PRODUCTION POSSIBILITIES FOR ENERGY[1]

This appendix gives long-term estimates of the possibilities for different types of energy. Three groups of energy sources are considered in turn, fossil energy, fissile energy and flow energy. The time horizon for the depletable types, the first two, is set at the comfortably long period of 10,000 years. Flow energy will last as long as the sun. The results of the analysis are given in table A.2.5.

1. Fossil reserves

Estimating reserves of energy resources (or any other non-renewable stocks such as ores) is not a matter of hard scientific measurement. Publicly accessible data on estimated exploitable reserves[2] at any moment in time depend on at least five factors:

- ◇ current and expected market prices
- ◇ explorations done
- ◇ available technologies for exploitation and transport
- ◇ public availability of known facts
- ◇ total amounts available on earth

The latter notion indicates the "real reserves". The notion of "ultimate reserves" indicates the part of the total amount available that eventually could be extracted. The estimated reserves are usually speculative or even unknown. The first three economic factors determine the ratio between these speculative real reserves and the exploitable reserves. These first three factors are clearly not constants. Thus, long-term estimates of reserves have to consider the variability of these factors. Some external variables have prime importance. If demand grows faster than supply prices go up. The three factors relate dynamically to each other. If prices go up explorations go up. If costs of available technologies go down exploration goes up. If the market becomes less oligopolistic exploration goes up but long-term development in technologies may slow down. Given a minimum degree of competition, extra exploration will lead to extra production, thus pressing down prices. If real limits in availability on earth are reached prices will go up, not to much avail. Finally, even if data are known to explorers, they may have good reasons not to make them public. What developments have taken place with respect to the factors concerning fossil energy resources and their estimates?

Price

Oil has been the leading reserve in the last decades, replacing coal in that function. Apart from peaks in the Seventies ending in the beginning of the Eighties, the long-term price of oil has been remarkably constant in the last three quarters of a century, keeping in mind that amounts produced have increased tremendously. Its price has even had a long-term tendency to decline. Between 1950 and 1970 prices halved, in real terms, from around $10 to around $5 per barrel of 159 l (in 1974 $, Odell 1990, p.7). In the Seventies the price exploded to a nearly tenfold level reaching its all-time high in 1981. After that year the long-term price erosion set in again, with prices in 1992 between $15 and $20 per barrel. The price rise of the Seventies led to the development of large scale offshore exploitation with cost prices of around $15 per barrel. The rise in conventional reserves has taken place in the context of these rather stable long-term prices.

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1 Ruben Huele, CML, aided in the production of this appendix.

2 This is a somewhat broader category than "proven exploitable" but more restrictive than "estimated stocks" or "ultimate stocks".

Exploration
The possibility of a long-term cartel keeping prices up and politicizing supply gave strong incentives to new exploration outside OPEC countries. However, at a global level, exploration is very limited still. Odell (1990) gives a global survey of "potentially petroliferous areas" and the amount of drilling done. Potentially petroliferous areas are quite evenly spread around the world, mainly on the edges of all continental shelves. Wells are not. In the US several million wells have been drilled. In the Soviet Union this number is about half a million and in Canada two hundred thousand. The total for the rest of the world is less than half a million, including the Middle East. The costs of exploration have declined sharply because of better models and software for the interpretation of seismic experiments.

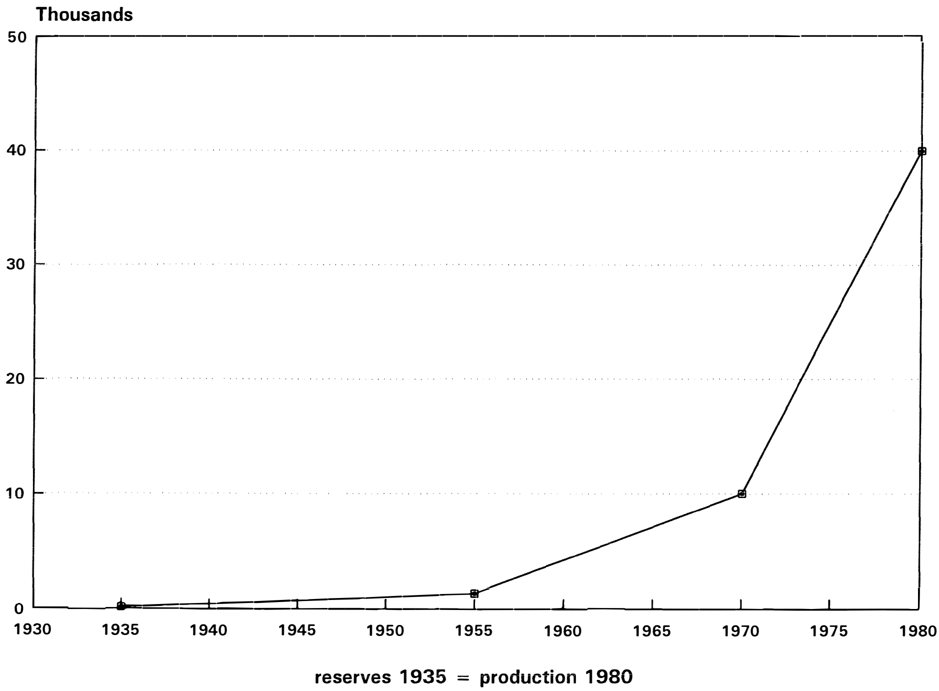
Technologies
The technologies for exploitation and transport developed in this period now make exploitation worthwhile in hitherto closed or unprofitable surroundings. Moreover, drilling technologies allow the drilling of deeper wells than previously had been possible. The costs of exploitation have declined because of "horizontal drilling", a technique that allows the extraction of oil and gas from larger areas through one main pipe. From one site the amount of oil that can be cost-effectively extracted is usually less than half the amount present, sometimes not more than ten percent. The exploitable reserves in known sites thus have been increased substantially by the lowering of costs made possible by horizontal drilling.

Historical reserves
Estimated and proven exploitable reserves tend to grow in time, with additions to reserves far exceeding their use. In the forty-five years between 1935 and 1980 global reserves of crude oil grew by a factor of three hundred. That is an average of over eleven percent per year. This growth in reserves is not yet slowing down. If any trend may be discerned it is an acceleration of growth, see figure A.2.1 below. Annual use is now as high as total estimated reserves were in 1935. In the Seventies, the growth rate for reserves of natural gas was far exceeding that of oil, at twelve percent per year for oil and at twenty three percent for natural gas. The ratio of reserves to yearly use has increased substantially for gas and slightly for oil in the last decades. The mechanism responsible for this usually stable amount of reserves can be primarily related to the number of producers, and not to actual or potential reserves. If there were one producer only, it would not make sense for him to go on exploring if his proven reserves would last him for more than fifteen years. The interest on the cost of this too early exploration would be lost. His proven reserves coincide with total proven reserves. If there are many firms, these might all want to increase their market share and thus might quite rationally explore more extensively, but not for a longer time horizon per producer. Thus, historically, the amounts of reserves may be based more on the number of producers and their aims than on any scarcity-related underlying reality[1].

Future reserves of conventional fossil reserves
How can an estimate of long-term fossil reserves be made? Seven widely differing methods can be employed. First, the rising trend in the ratio between reserves and yearly production could be extrapolated. This model is easy and fits best with historic reality. It supposes, however, that in the real long term the whole earth, and even more, is available in the form of oil. Any future depletion is modelled away. This is not acceptable for the analysis here.

[1] This mechanism might explain why the proven reserves of oil in terms of number of years at "current" consumption levels slightly increased over the last decades, when non-OPEC producers increased their market share. The return to a higher share of OPEC now expected to take place in the next decades would then lead to a decreasing amount of proven reserves in terms of years of current exploitation.

FIGURE A.2.1 THE GROWTH IN TIME OF PROVEN EXPLOITABLE RESERVES OF CRUDE OIL



Secondly, a constant ratio of usable reserves might be supposed, which still models away any possibility of future depletion.

Thirdly, current growth rates of reserves could be extrapolated. Figure A.2.2 does this on a logarithmic scale to the year 2020. Again, the possibility of depletion is nearly impossible with such a method. Only an even faster increase in demand could lead to depletion.

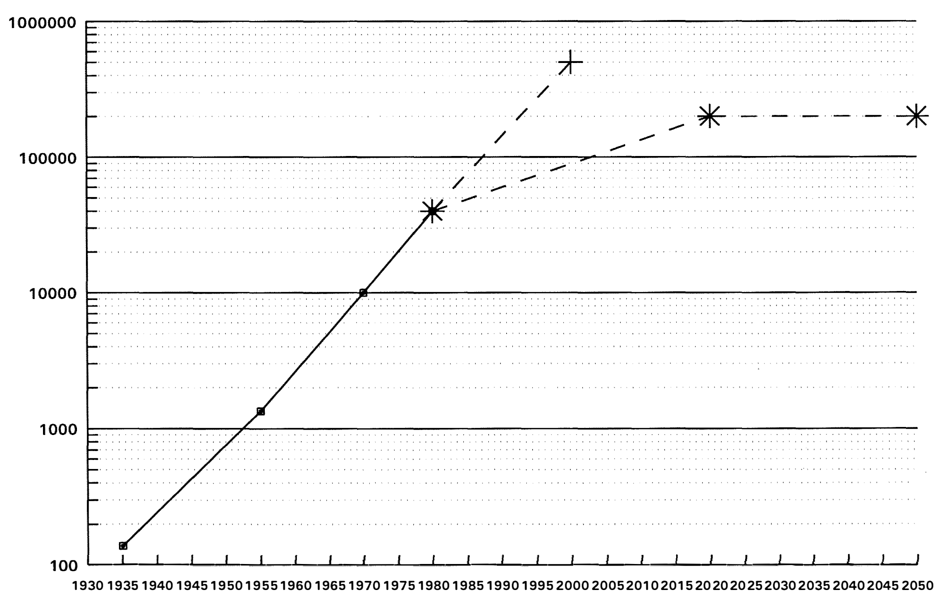
Fourthly, the latest available estimate of reserves could be used, assuming no new reserves will ever be found. This makes no sense either, as it contradicts the turbulent growth in reserves currently taking place.

A fifth, more subtle approach would define reserves as a fraction of geological estimates ("ultimate resource in place") in relation to the price. Suppose a certain price rise will occur and the reserves can be computed. If there is no limit on prices, there is no limit on resources in this type of model either. The basis for such an exercise is small in terms of available data. See de Vries (1991).

The sixth method is the most serious one for fossil energy resources, but is not available practically. Hubbert (1969) applied it in the most thorough study on reserves available. The basic idea is that a relation may be found between the amount of drilling, in metres, and the amount of oil (or gas) discovered, as a function of time. For the US this has been done. Cumulative production lags behind cumulative discoveries by a bit over a decade. If the amount of discovery per metre of drilling declines, the growth of discoveries declines. With a time lag of about half a decade reserves will start to decline. In the US this clearly is the case. Exponential growth has come to a halt and, through use, reserves are declining now. Technological development may slow down the decline a bit but the structural trend is clear. Since most drilling has occurred in the US, much more than in the rest of the world together, the extrapolation of US data to world data is only possible on the basis of unfounded assumptions. I here make them. Suppose that relations between amount of drilling and amount of fuel discovered would have the same pattern as in the US. Suppose that, on average, the amount of oil per surface area of potential oil bearing geological formation is equal for all such potentially petroliferous areas in the US and the rest of the world. The US share (excluding Alaska) in such formations is about 5% (after Odell 1990). US cumulative ultimate discoveries (ex Alaska) are estimated by Hubbert at 1320 EJ. Adding a factor 2 for improved efficiency in recovery gives a US total of 2600 EJ. World reserves of oil then would be 52,000 EJ. The method has some drawbacks however. The main problem is that an exponential decline curve of discovery per metre of drilling is fitted to the empirical material. That material shows a quite erratic change in time. High yields per metre of drilling, from 1910 to 1945, quite suddenly diminished to about a quarter of the previous level and then remained stable or even increased. The best line clearly is neither linear nor logarithmic. The best fit is given by two straight lines, that both move more or less horizontally in time. If that best fit is accepted, however, there is no limit again on reserves still to be discovered.

Finally, a seventh method is to suppose that, after some time, the rise in reserves will come to an absolute halt. This conservative approach is applied here in estimating future reserves. For current purposes I assume here that rises in reserves will slow down in the next decades and will come to an absolute halt in the centuries to come, see the lower line in figure A.2.2 for the "estimated long-term reserves" of oil. The assumption of an absolute stop on the discovery of new exploitable reserves makes it a conservative estimate. With coal, crude oil and natural gas this approach is used here. There are, however, very large reserves of other, mostly unused fossil fuels for which this approach is not possible. These are often called the non-conventional reserves. Some separate estimates of these reserves have been included.

FIGURE A.2.2 RESERVES OF OIL. HISTORICAL DEVELOPMENT, EXTRAPOLATION, AND CONSERVATIVE LONG-TERM ESTIMATE (*), IN EJ, LOGARITHMIC SCALE



reserves 1935 = production 1980

Reserves of non-conventional fossil reserves[1]

-Ultra heavy oil

This oil has to be dispersed before it can be pumped. In Venezuela, for example, there is a layer of this oil at a depth of a few hundred metres, stretching over an area of around three hundred by one hundred kilometres. It is called the Orinoco Belt. The exploitable amount of this ultra heavy oil can only be guessed. It is somewhere between the Middle East oil reserves and world coal reserves, that is roughly 250,000 ExaJoules[2]. Production has started in Venezuela at a modest amount of half a million barrels a day, at production costs of around $5 per barrel (personal communication Odell). British petroleum is marketing this quite contaminated product in Europe.

-Tar sands

Tar sands are an option that is technically possible. In exploitation it is too costly to compete in current energy markets. The amounts are large. One very conservative estimate is included.

-Shale oils

Shale oils are a most abundant resource of fossil energy. Technically, production is possible now. But cost estimates range from $30 to $150 per barrel (personal communication Odell). At current market prices exploitation and thus exploration is not feasible economically. One estimate is included, but not in the total estimates.

TABLE A.2.1 ESTIMATES OF GLOBAL FOSSIL ENERGY RESERVES, IN EJ ($=10^{18}$J) THERMAL, GROSS*

PRIMARY SOURCE	1. ESTIMATED RESERVES 1969[3]	2. ESTIMATED RESERVES 1980[4]	3. LONG-TERM ESTIMATE OF RESERVES
coal	250,000	420,000	700,000
crude oil	10,000	40,000	200,000
natural gas	10,000	110,000	300,000
non-convent. gas	?	?	~1,000,000
ultra heavy oil	?	?	250,000[5]
tar sands	»2,000	50,000	50,000
shale oil	2,000,000	2,000,000	2,000,000
fossil total (ex non-conv. gas and shale oil:)	**2,272,000 (272,000)**	**2,620,000 (620,000)**	**4,500,000 (1,500,000)**

[1] There are indications that natural gas is formed from the core and mantle of the earth, "bubbling" out form under the edges of the continental shelves and seeping through cracks. In that case natural gas would partly be a flow entity of an as yet unknown amount.

[2] All figures, including those in the tables, are in exaJoules, that is 10^{18} Joule. 1 ExaJoule is equivalent to about 25 million tonnes of oil.

[3] Hubbert (1969) From: J. Harte (1988)

[4] Skinner (1986), WRI (1987), From: F.Langeweg Ed. (1989)

[5] Own estimate, partly based on personal communication with P.R. Odell.

-Gaseous salt brine
Non-conventional gas reserves are extremely large. In deep layers of brine estimates go as high as 110,000 exaJoules (Andriesse 1991). The first production started in the USA. One main problem is to get rid of the brine. These data have not been included in the total estimates.
-Gas in shales, sandstones and coal layers
In shales, sandstones and coal layers there is an estimated exploitable reserve of natural gas of 4,000 exaJoules (Andriesse 1991).
-Non-conventional gas, in clathrate structures
By far the largest and most concentrated reserves, however, are in watery-icy solutions, the gaseous clathrate structures. These lie no more than a few hundred metres deep under most seas, and in permafrost areas. Estimates are up to 800,000 exaJoule (Appenzeller 1991). Techniques for exploitation are not yet available but do not seem excessively complex. These estimates of non-conventional gas reserves are enough to cover several thousand years of total current fossil energy use. They have not been included in the tables.

Long-term supply and its consequences
If the price constraint on exploration is relaxed and levels in the order of $100 are acceptable (that is about $0.60 per litre) the amounts of conventional fossil reserves can cover at least five thousand years of current energy use[1]. Including shale oil, a production equal to current total energy use may be realized for far more than ten thousand years. Adding non-conventional sources of natural gas would add several thousand years of current levels of consumption.

TABLE A.2.2 ESTIMATES OF WORLD ENERGY USE OF FOSSIL RESOURCES AND OF POTENTIAL VERY LONG-TERM FOSSIL ENERGY PRODUCTION, IN EJ ($=10^{18}$J) THERMAL, GROSS*

PRIMARY SOURCE	LONG-TERM ESTIMATE OF RESERVES	USE 1980[1]	POTENTIAL PRODUCTION PER YEAR OVER 10,000YRS
coal	700,000	90	70
crude oil	200,000	135	20
natural gas	300,000	60	30
non-conventional gas	~1,000,000		~100
ultra heavy oil	250,000	>0[2]	25
tar sands	50,000	>0	5
shale oil	2,000,000	0	200
fossil total **(ex non-conventional gas and ex shale oil)**	**4,500,000** **(1,500,000)**	**285** **(~0)**	**450** **(150)**

[1] At that price, demand for fossil energy resources would of course drop substantially through energy saving and energy source substitution.

[2] Current production is 0.5 million barrels per day. Production costs are around $5. Personal communication P.R. Odell.

2. *Fissile reserves*

Estimates of reserves of fissile resources are similarly time dependent. The costs of exploration are wasted if known exploited reserves may serve current use for decades. In the current situation, with demand behind former predictions, geological reserves are exploitable only if rail or water transport is readily available (personal communication Kluijver, a former chief geologist at a larger exploration firm). Current estimates are thus hardly indicative of the amounts that may be extracted in the long run. Without breeder reactors there is only an insignificant contribution of fission power to long-term energy consumption, if the extremely conservative estimates of reserves of uranium are accepted as truth. Economically and technically it is very possible that at higher energy prices and with development in extraction technologies, exploitable reserves will increase by several orders of magnitude. With the use of current fission technology only, nuclear energy could already give a substantial long-term contribution.

TABLE A.2.3 ESTIMATES OF WORLD RESERVES AND OF LONG-TERM POSSIBILITIES FOR NUCLEAR ENERGY PRODUCTION, IN EJ ($=10^{18}$J) THERMAL, GROSS*

PRIMARY SOURCE	ESTIMATE OF RESERVES 1969[1]	LONG-TERM ESTIMATE OF RESERVES	POTENTIAL PRODUCTION PER YEAR OVER 10,000YRS
uranium for convent. nuclear reactors	20,000	50,000	5
uranium and thorium for breeder reactors	10,000,000	10,000,000	1000
deuterium and lithium for fusion reactors	(10^{12})	(10^{12})	not counted
total, ex fusion	**10,020,000**	**10,050,000**	**1,005**

If in the long run the breeder technology should be used, the reserves of fissile material are nearly inexhaustible. Thorium is as abundant in the earth's crust as lead, mostly combined with other economically attractive minerals. Current estimates of fissile reserves are based mainly on general geological knowledge. Specialized exploration, aided by satellite surveys, probably will lead to an explosive growth of fissile reserves. However, even with reserves as estimated in 1980 current energy consumption could be fully served for another thirty thousand years[2], using the breeder technology. This, however, would create a very large stockpile of nuclear waste. Also, calamities could ruin some hospitable parts of the world.

The technology for fusion power has proven extremely awkward to develop. Capital costs will probably be very high, and nuclear waste will be produced in large quantities. The possible future contribution of fusion power has been disregarded altogether.

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[1] Hubbert (1969) From: J. Harte (1988)

[2] It is not clear from the sources used which definition for reserves is applied. The United Nations Energy Statistics definition expresses nuclear power as the heat value of the electricity produced. The industry tends to express it as the heat produced by fission. The latter definition is assumed here and seems to be the correct one. It is used by protagonists of nuclear power. If figures were based on the UN definition they would indicate roughly twice the number of years of possible fissile energy use.

3. *Flow energy*

Flow energy at present has a very limited part in energy production of only a few percent. If necessary it could be expanded substantially within decades, based on current technologies, especially for photovoltaic cells. The costs of most forms of flow based energy are still higher than those based on polluting fossil energy resources. Several types of flow energy may contribute

◇ direct transformation of solar radiation into electricity with photovoltaic cells and with steam production using mirrors; in hydrogen, through hydrolysis; and in passive low-caloric heat
◇ power production with indirect solar energy as electricity, with water and wind turbines
◇ biomass production with solar radiation, in agriculture
◇ earth heat, usually by pumping low or medium caloric water from underground reservoirs

Each of these alternatives will now be investigated in turn, starting with a survey of general data on the earth as used in the analysis.

General data for flow energy

Surface of the earth	510 * 10^{12} m^2
Land area	150 * 10^{12} m^2
Surface of agricultural soils	50 * 10^{12} m^2
Surface of tropical deserts	40 * 10^{12} m^2

Direct solar energy, photovoltaic (=electric) and hydrogen

5% of tropical deserts (= 0,7% land area)	2. * 10^{12} m^2
Inflow of solar energy in desert areas	7.6 * 10^9 J/m^2 year
Inflow of energy in solar cells	15. * 10^{21} J/year
Average efficiency of solar cells	10%
Electricity/hydrogen yield	**1,500. * 10^{18} J/year**
Passive solar energy	**p.m.**

* Placing solar cells in deserts has as advantage that solar inflow is relatively high and regular, and that alternative costs are low. Also, some current oil producers could invest in energy diversification from current proceeds of fossil energy resources. The disadvantage is the longer transport distances required.

* There now are solar cells with an energy efficiency of 30%; cheaper versions on the market have an efficiency of 7%; solar powered steam cycle power production with an efficiency of 12 % and hydrogen producing cells with an efficiency of 9%. The stated long-term efficiency of 10% is thus a conservative estimate for the long term.

* The energy yield is not fully equivalent to the figures for fossil energy production. These are in terms of thermal energy, while solar power produces electricity directly. Part of fossil energy is transformed into electricity with a long-term efficiency not much higher than 50%. The comparable solar energy figures would be somewhat higher than stated.

* Passive solar energy has not been quantified. Yields could be substantial since the transformation efficiency is extremely high, in the order of 90%. however, transport of this low-caloric energy is hardly possible.

Indirect solar power: hydro power and wind power

Current use in the order of	6. * 10^{21} J/year
Inflow of solar energy in tropical seas:	
Area 15% of total sea surface	70. * 10^{12} m^2
Inflow of energy per square metre	7.6 * 10^9 J/m^2 year
Energy to air, evaporation + convection	530. * 10^{21} J/year
Energy of water + wind to land, 50%	250. * 10^{21} J/year
Suppose 1 promille effective, **yield (J_{elec})**	**250. * 10^{18} J/year**

* Constant sea temperature assumed
* Energy to air exclusive of terrestrial heat to seas
* One promille is an unfounded estimate. A high portion of wind energy would slow down wind on earth. The structures required for such an endeavour seem hardly possible. The potential energy of rain is now used in amounts of more than an order of magnitude less then one promille. An increase of hydroelectric power by a factor of ten would imply a huge transformation of ecosystems. The deforestation usually implied would release large amounts of carbon dioxide and methane.

Biomass

Three alternative computations for biomass production are given, Dutch potatoes, grasses in wetlands, and one percent of global biomass fixation.

Biomass for energy production, thermal, Dutch potatoes

Energy content	4.44 * 10^6 J/kg
Yields 35 tonnes per hectare	3.5 kg/m^2.year
Energy yield	15.5 * 10^6 J/m^2.year
10% of global agricult. area	4.6 * 10^{12} m^2
Gross energy yields	**71.0** * **10^{18}** J/year

* Energy content is the caloric value at digestion.
* Energy input in agriculture is substantial. In the Netherlands there is a negative efficiency for the whole of agriculture, including greenhouse production.
* Figures are gross values, energy requiring inputs have not been subtracted.
* Experimental crops such as the reed *Miscanthus sinensis* could have yields in the order of 30 tonnes of dry biomass (Dunn et al. 1992).

Biomass for energy production, thermal, grasses in wetlands

Energy yields	30. * 10^6 J/m^2.year
10% of global agricultural area	4.6 * 10^{12} m^2
Gross energy yields	**140.** * **10^{18}** J/year

* Energy yields are based on the energy content of the carbon fixed.
* Estimates are based on prime productivity of ecosystems. Wetland grasses have the highest prime productivity of all ecosystems
* Harvesting would require the usual agricultural inputs. These have been omitted.

Biomass globally

Total primary production	0.5 * 10^{15} kg C / year
(Vitousek et al. state 0.494; Harte 0.55)	
Energy content	35. * 10^{6} J / kg
Energy fixed, total	17. * 10^{21} J / year
Available for energy production, 1%	10^{-2}
Total energy from biomass	**170. * 10^{18} J / year**

* 2.2 kg of biomass = 1 kg of carbon
* Transformation efficiency (solar radiation on soil to biomass on soil) of plants around 0.5%
* One percent is an unfounded estimate to assess the potential order of magnitude of energy farming.

Wastes from agricultural produce ***p.m.***

* The amounts are large. For many crops these wastes are used already for energy production or for some other function. It is unclear if a substantial increase of use of agricultural wastes would result in a net energy contribution. Converting wastes into fodder, or even food, might be a better contribution to societal energy efficiency.

The different estimates of biomass production seem to indicate the same order of magnitude. Energy farming on the scale implied will have serious effects on the possibilities for nature and for food production. Therefore, a low estimate is given for biomass production.

Total biomass:

Estimate	**100. * 10^{18} J/year**

Potential of terrestrial heat ***p.m.***

Total amounts of potential flow energy

Direct solar energy	1,500. * 10^{18} J/year
Indirect solar energy	250. * 10^{18} J/year
Biomass production	100. * 10^{18} J/year
Passive solar energy	p.m.
Terrestrial heat	p.m.
Total	**1,850. * 10^{18} J/year**

* Energy required for producing secondary energy is not subtracted; gross figures are given, also for the other types of energy. An intuitive estimate is that the difference between gross and nett are
 # smallest for passive solar energy
 # medium for hydro and wind energy
 # medium for fossil energy (including energy for emission reduction)
 # medium for solar cells
 # highest for fissile energy and biomass production

Without any restriction in time, flow energy may be used at levels of six times current total energy use. Main source is the conversion of direct solar energy. For the location tropical and subtropical deserts would be prime choice. If the conversion efficiency of direct solar energy increases to the levels currently approached on laboratory scale, the potential rises by a factor of 3. Biomass production has a gross conversion ratio more than an order of magnitude lower than that of direct conversion. The factor to be subtracted to arrive at net production is much higher for biomass production than for the other types of flow energy. Production requires energy for fertilizers, pesticides, soil and produce handling and transport. Moreover, biomass production will partly compete with food production.

TABLE A.2.4 ESTIMATES OF LONG-TERM ENERGY PRODUCTION FROM FLOW ENERGY, IN EJ ($=10^{18}$J) THERMAL, GROSS*

PRIMARY SOURCE	USE 1980[1]	POTENTIAL PRODUCTION PER YEAR, PERMANENTLY[1]
solar energy: electric/hydrogen/low caloric	>0 (partly MJelec)	1,500 (partly MJelec)
biomass	12[2]	100
water power and wind power	MJelec 6	MJelec 250
earth core heat	?	p.m.
flow energy total	>18	**1,850**

4. Conclusions on the depletion of energy

The combined use of the three main types of energy together allows for an energy consumption of ten times current consumption for ten thousand years, based on the still conservative estimates as given above. The main conclusion here is that there is *no problem of energy depletion*. Use of the limited regulating capacity of environmental agencies for the prevention of energy depletion is a *waste*.

The facts used in this section are not new nor based on complicated research. The facts are reasonably well known, if presented in a slightly different way. It seems odd that such facts do not find their way to policy-makers and environmentalists[3]; that proposals for policies aimed at depletion prevention are still developed.

[1] See the appendix to this chapter for the basic assumptions and computation.

[2] Nearly exclusively as fuel wood, see substance flow scheme hereafter. Assumptions: Carbon 80% of dry weight of biomass; 16 MJ per kg of dry biomass.

[3] This lack of connection between science and policy was noted by Tickell (1977) when researching the literature on climate change in the Seventies.

TABLE A.2.5 ESTIMATES OF TOTAL ENERGY RESERVES AND POTENTIAL LONG-TERM ENERGY PRODUCTION, IN EJ ($=10^{18}$J) THERMAL, GROSS

ENERGY SOURCE	Long-term ESTIMATE OF RESERVES	POTENTIAL PRODUCTION PER YEAR OVER 10,000YRS
fossil energy	4,500,000	450
nuclear energy	10,050,000	1,005
flow energy	(not applicable)	1,850
sum total		**3,305**

APPENDIX 5.5.3
FLOWS OF CARBON THROUGH THE ECONOMY:
STRUCTURAL CONTRIBUTIONS OF HUMAN ACTIVITIES TO THE LONG-TERM ATMOSPHERIC CARBON CYCLE
Results of a preliminary analysis[1]

The structural contributions to the carbon cycle are those that add to the cycle by "chemical forming" and by inflow into the economy by "extraction from the lithosphere", or that subtract from the cycle by "chemical destruction" (by irreversibly binding carbon) and by "deliverance to the lithosphere". Inflows from the environment" and "emissions to the environment" include biological flows.

The main flows may be grouped around these types of inflow or outflow and the main activities concerned. These are: production and use of fossil fuels (1), economic use of limestone (2), and all flows connected with vegetable and animal production such as food and wood (3). Several other large scale human interventions in the global carbon cycle are also discussed (4). These subflows are first be treated separately, with assumptions and references specified. Finally, they are brought together in the general substance flow table and scheme, in the main text, in section 5.5.4. The result of the flexible response strategy developed there is presented in detail in the final part of this appendix (5).

1. Production and use of fossil fuels
Fossil fuels (oil, coal, and natural gas) are extracted from the lithosphere for use in the economy. The bulk is used in combustion processes. Some of the fossil fuels are used as starting materials in the chemical industry. If combustion is complete, all the carbon in fossil fuels is transformed into CO_2. In practice a small part of the carbon is freed in the form of soot(particles). Furthermore methane is released in the extraction, transport and use of fossil fuels.

1.1 Extraction and transport of fossil fuels
In 1987 6052 * 10^9 kg C was extracted from the lithosphere in fossil fuels (table A.3.1). As a result of combustion processes 5911 * 10^9 kg C from that was directly emitted in the form of CO_2. The combustion of fossil fuels in OECD countries (World Resources, 1990) can be related to several types of functional application:

- 45% industry
- 35% commercial and residential
- 20% transport

Until now the all the CO_2 formed in the combustion process is emitted from the economy to the environment.

Losses at extraction

The losses at extraction of coal and oil are estimated at 1%. The losses at gas production are given under "methane emissions". Any amount of oil and coal lost is transformed into CO_2 in the end. Therefore the losses form an indirect CO_2-flow from the economy to the environment:

oil :	24	
coal :	23	+
total:	48	

[1] This appendix is the co-product of the author, E.v.d. Voet and E.G.M. Kleijn. The results, including mistakes, are my responsibility.

TABLE A.3.1 PRODUCTION OF FOSSIL FUELS[1]

	production (PJ)	energy-content (MJ/kg)	production (10^9 kg or m^3)	emission factor (kg CO_2 /kg or /m^3)	extracted (10^9kg C)	other use	combustion of C[2] (10^9 kg)
Liquid	123175	42	2933	3.1	2432	50	2382
Solid	91091	30	3036	2.9[3]	2362	8	2354
Gas	66696	32	2084	1.8	1209	0	1209
Peat[4]			21		2	0	2
Total					6005	58	5947

Methane emissions

At extraction and in the transport of all fossil fuels methane emissions occur. These are flows from the economy to the environment. From measurements of carbon isotope abundance in methane in the atmosphere the ratio of methane from fossil origin to that of current biological origin can be computed. Based on this ratio and estimated total emissions of current biological origin, the total emissions of methane from fossil sources is estimated to be 118 $\pm$ 20 * 10^9 kg C (Lelieveld and Crutzen, 1991). About 20% of these emissions cannot be accounted for:

- gas drilling, venting, transmission	45	(IPCC, 1990)
- coal mining	35 $\pm$ 10	(Lelieveld)
- oil drilling	14 $\pm$ 6	(Lelieveld)
- other	24 $\pm$ 4	(Lelieveld)
	118 $\pm$ 20 * 10^9 kg C	

1.2 Non-combustion use of fossil fuels

Other types of use are about 50*10^9 kg C from oil[5] in the production of plastics and about 8*10^9 kg C from coke in the production of steel[6].

Of the 50*10^9 kg in plastics about 10% is assumed to be used in applications with a short life span. In OECD countries about 40% of the waste is incinerated (OECD, 1991). On a world scale the amount of waste which is incinerated is estimated at 20% and the amount dumped at waste disposal sites is estimated at 80%. Therefore about 1*10^9 kg C is burned and flows as CO_2 from

---

1 *production : World Resources Institute (1992); data 1987
*energy-content : Gool et al. 1987, further referred to as "Poly-energy pocketbook".
*emissionfactors : CBS 91/1 ; Dutch data
*other use : 3% oil for plastics, derived from world production plastics (TNO, 1986).

2 Assuming 1% losses during extraction and transport of oil and coal. For natural gas, the total loss is estimated by the IPCC (1990) to be 34*10^9 kg C.

3 Emission factors of CO_2 for combustion of coal in kg CO_2/kg coal:
hard coal: 2.5-3.3
coke: 3.0-3.6
lignite: 2.1
The average is estimated at 2.9.

4 Peat for fuel use only. For assumptions and calculations, see section 3.1.

5 The world production of plastics amounts to about 60*10^9 kg C. When a high estimate is used for the carbon content of these plastics (86%) the total amount of carbon in these plastics is about 51*10^9 kg.

6 The world production of crude steel amounts to about 777,784*10^6 kg. The average carbon content is 1% therefore the amount of carbon incorporated in steel amounts to about 7.8*10^9 kg C.

the economy to the environment and about $4*10^9$ kg accumulates[1] in the economy. Plastics used in durable applications accumulate in the economy.

The amount of C incorporated in steel accumulates in the economy, either in the applications themselves or on the waste disposal sites.

Accumulation in the economy:	
plastics in applications	45
plastics on dump-sites	5
steel in applications	7
steel on dump-sites	1

	58
Emission as CO_2 :	
plastic incineration	1
plastic decomposition	0
steel oxidation	0

	1

2. Economic use of limestone

A survey of totals is given in 2.3.

2.1 Production of CaO from limestone

Limestone is used in the production of cement and some types of plaster, and in CaO production for industrial applications. It also is used, as a carbonate, in building material, roads and agriculture.

2.1.1 Cement

One kilogramme of cement contains 0.64 kg CaO. The calcium fraction in CaO is 0.71. The calcium fraction in cement is 0.457. The limestone transformed has one carbon atom for each calcium atom. The amount of carbon freed per kg of cement is 12/40*0.457 = 0.137 kg.

<u>Chemical forming and emission</u>

Cement production:	Neth.[2]:	World[3]:
Portland	$2.613 * 10^9$ kg	$450 * 10^9$ kg
Furnace slag	$3.201 * 10^9$ kg	$449 * 10^9$ kg
Total	$5.814 * 10^9$ kg	$899 * 10^9$ kg

At chemical forming all carbon is emitted to the environment.

Carbon emissions at production	Neth.:	World:
Portland	$0.358 * 10^9$ kg	$61.7 * 10^9$ kg
Furnace slag[4]	$0.439 * 10^9$ kg	$61.6 * 10^9$ kg
Total	$0.797 * 10^9$ kg	$123.3 * 10^9$ kg

---

[1] Assuming that the plastics do not decompose on dump sites.

[2] All data for the Netherlands from Wolf (1990).

[3] Chemie Kompendium, 1986. Data of 1984 have been used to estimate data for 1987.

[4] Composition assumed similar to that of Portland cement. In both types of cement the contribution of gypsum to calcium content is disregarded.

Extraction from the environment and chemical destruction

Carbonation of	Neth.:	World:
all concrete, 10%[1]	79.7 * 10^6 kg	12.3 * 10^9 kg

Net emissions from		
cement applications:	Neth.:	World:
Total	0.719 * 10^9 kg	111.0 * 10^9 kg

2.1.2 Plaster

Chemical forming and emission

CaO production, net	Neth.:	World:
of 10% calcination	0.732 * 10^9 kg	- -

Carbon emissions		
at production	Neth.:	World:
Total	0.157 * 10^9 kg	24.2 * 10^9 kg[2]

2.1.3 Industrial applications

Chemical forming and emission

Industrial applications mainly use CaO.

CaO production,	Neth.:	World:
total industrial	0.178 * 10^9 kg	- -

Carbon emissions		
at production	Neth.:	World:
Total	0.04 * 10^9 kg	0.62 * 10^9 kg[3]

2.1.4 Extra weathering in carbonate applications

Chemical forming and emission

	World:
Buildings (sandstone)	p.m.
Roads	p.m.
Agriculture	p.m.

[1] It is assumed that ten percent of all calcium in cement produced will be carbonated. The ten percent is deducted from current production instead of current stock. This leads to an overestimation of carbonation since cement production is rising.

[2] The Dutch proportion of emissions from cement to those of plaster and global data on cement are used to estimate the global emissions from plaster.

[3] The Dutch proportion of emissions from cement to those of plaster and global data on cement are used to estimate the global emissions from plaster.

2.2 *Use of limestone in binding sulfur*

Limestone is used in several techniques to bind sulfur, freeing the carbonates in the process. Desulfurisation is mostly applied at the incineration of fossil fuels. Data exclude the carbon from the fuels themselves. Coal and oil are the main sources of sulfur. After chemical forming the carbon is immediately emitted as CO_2. All data are from 1987.

Chemical forming and emission

2.2.1 Coal

The set-up of the calculations is as follows:
*of the global coal production 50% is used in large furnaces (own estimate)
*sulfur content is 2.5%[1]
*30 % of all large furnaces use desulfurisation installations (own estimate)
*desulfurisation efficiency is 80% (own estimate)
*each atom of sulfur bound frees one atom of carbon, as CO_2.

Data:

Coal production	$3113.7 * 10^9$ kg[2]
Sulfur bound	$9.3 * 10^9$ kg[3]
Carbon freed	$3.5 * 10^9$ kg[4]

2.2.2 Oil

The set-up of the calculations is as follows:
*oil used in large furnaces is 25 % of coal used in large furnaces, is 12,5% of all coal[5]
*sulfur content is 1.5%
*30 % of all installations use desulfurisation installations (own estimate)
*desulfurisation efficiency is 80% (own estimate)
*each atom of sulfur bound frees one atom of carbon, as CO_2

Data[6]:

Oil in large furnaces	$389.0 * 10^9$ kg
Sulfur bound	$1.4 * 10^9$ kg
Carbon freed	$0.5 * 10^9$ kg

2.3 *Total flows from use of calcium carbonates*

	Chem. form.	Extr. env.	Emission
Cement	$123.3 * 10^9$ kg	$12.3 * 10^9$ kg	$123.3 * 10^9$ kg
Plaster[7]	$24.2 * 10^9$ kg		$24.2 * 10^9$ kg
CaO, industry	$0.6 * 10^9$ kg		$0.6 * 10^9$ kg
Extra weathering of carbonates in applications: p.m.			
Desulf. coal + oil[8]	$4.0 * 10^9$ kg		$4.0 * 10^9$ kg
Total	**152.1** $* 10^9$ kg	$12.3 * 10^9$ kg	$152.1 * 10^9$ kg

---

1 Harte (1988): 2.5%; Chemie Kompendium (1986): 0.5 - 2.0%; Polyenergy pocketbook (Dutch): 0.2 - 10%. 2.5 percent is their own estimate, based on the fact that high sulfur coal will be burned more in larger installations.

2 Polyenergy pocketbook p.864, data 1984.

3 Polyenergy pocketbook p.864, data 1984.

4 Polyenergy pocketbook p.864, data 1984.

5 OECD Environmental data compendium (1991)

6 Polyenergy pocketbook p.864, data 1984.

7 The Dutch proportion of emissions from cement to those of plaster and global data on cement has been used to estimate the global emissions from plaster.

8 Polyenergy pocketbook p.864, data 1984.

3. Vegetable and animal production

With the production of biological material, CO_2 is extracted from the environment. The CO_2 extraction in fodder, food, and wood is quite substantial, even as compared to fossil fuel extraction. However, for the most part this carbon is emitted into the environment again within a year, through oxidation in the economy or as waste products to be oxidized in the environment. Some of these emissions are more harmful from a greenhouse point of view than CO_2. There then is a net positive greenhouse effect, as in the formation of methane from organic material. Only a minor part is added to economic stocks, mostly in lasting applications of wood.

3.1 Agriculture

The extraction of carbon from the environment consists of agricultural products: grains (including wheat, barley, maize and rice), grass, and "others" (including soy, potatoes and sugar). Only the carbon in harvested parts is considered; roots and straw are considered to remain in the environment. Fodder used for animal production is included, animal production itself is placed in the economy. Only fish and game are considered extractions from the environment. Human and animal waste products add up to the total emission of the food production sector. These include all production losses during food processing, etc., which has not been specified separately. Specifications are given below.

Economic inflow from immobile stocks

In this category there are two items: the extraction of peat for agricultural use, and the oxidation of previously "immobile" organic matter in soils.

For peat extraction, this amount is calculated as follows: peat extraction[1] * 0.36[2] * 0.32

peat: 168,121 * 10^6 kg * 0.36 * 0.3 = 18 * 10^9 kg C

Oxidation of organic matter in soils, i.e. transformation of organic C into CO_2: normally, this oxidation is balanced by a renewed inflow of organic matter. In situations where "immobile" organic matter is brought into the cycle by deep ploughing, there is a net inflow to the economy, with a simultaneous emission to the environment (see below).

Extraction from the environment

Extraction is solely in the form of CO_2 fixation in organic matter, calculated as follows: parts harvested, by mass[3] * 0.36[4] * 0.3[5].

wheat:	510,462 * 10^6 kg * 0.36 * 0.3 =	55 * 10^9 kg C
barley:	180,357 * 10^6 kg * 0.36 * 0.3 =	20 * 10^9 kg C
maize:	451,086 * 10^6 kg * 0.36 * 0.3 =	49 * 10^9 kg C
rice:	465,780 * 10^6 kg * 0.36 * 0.3 =	50 * 10^9 kg C
Total		*174 * 10^9 kg C*
soy:	100,731 * 10^6 kg * 0.36 * 0.3 =	11 * 10^9 kg C
potatoes:	284,489 * 10^6 kg * 0.36 * 0.3 =	31 * 10^9 kg C
sugar:	96,639 * 10^6 kg * 0.40 * 1.0 =	37 * 10^9 kg C
Total		*78 * 10^9 kg C*
Total food and fodder, non-grass		**252 * 10^9 kg C**

[1] All data from UN Industrial Statistics Yearbook 1988, vol. II: Community Production Statistics, New York 1990.

[2] C content of dry organic matter and water content of fresh organic matter, see notes 4 and 5.

[3] Eurostat Agricultural Yearbook 1988 (1990).

[4] Carbon content of dry organic matter, based on formula $(CH_2O)_{106}(NH_3)_{16}(H_3PO_4)$ (from Harte 1988). For sugar the formula is $C_6H_{12}O_6$, leading to a carbon content of 40%. Data are presented in "white sugar equivalents", except for beet. For sugar beet a conversion factor is applied of 1/7.

[5] Own assumption: 70% of products consists of water. Sugar is assumed to contain no water.

Fish and game: only figures for fish are available. Game is assumed to be considerably less and therefore has been disregarded.

fish:	103,133 * 10^6 kg * 0.36 * 0.3 =	11 * 10^9 kg C

The extraction of grass is calculated as follows:
area[1] (ha) * grass production[2] (ton/ha) * 0.36 * 0.3

Africa:	787,934,000 * 0.5 * 0.36 * 0.3 =	43 * 10^9 kg C
Nth/C Am:	367,020,000 * 2.0 * 0.36 * 0.3 =	79 * 10^9 kg C
S Am:	472,777,000 * 2.0 * 0.36 * 0.3 =	102 * 10^9 kg C
Asia:	678,546,000 * 2.0 * 0.36 * 0.3 =	147 * 10^9 kg C
Europe:	83,979,000 * 5.0 * 0.36 * 0.3 =	45 * 10^9 kg C
USSR:	373,667,000 * 2.0 * 0.36 * 0.3 =	81 * 10^9 kg C
Oceania:	156,280,000 * 0.5 * 0.36 * 0.3 =	8 * 10^9 kg C
Total grass for fodder		**505 * 10^9 kg C**

In global cycle studies, the emission of methane from rice paddy fields is considered to be a major source. This emission, however, can be traced back to a prior extraction of CO_2 from the atmosphere. In terms of C, the extraction equals the emission mentioned below. In terms of GWP_{100} however, the emissions count for more than the extraction because of the 21 times greater warming potential of methane as compared to CO_2 in that period.

Emissions to the environment

The emissions consist of
-emissions directly from agricultural soil
-emissions from cattle
-human waste emissions.

Emissions directly from the soil are:
-the peat extracted from the immobile stocks, calculated as 18 * 10^9 kg C (see above)
-the emissions of methane from rice paddies. Sources vary from 19 * 10^9 to 128 * 10^9 kg C-CH_4, here the estimate used is 45 * 10^9 kg C[3].
-the oxidation of organic matter in the soil (see above: inflow from immobile stocks), no estimate.

Emissions from cattle are:
-methane emissions from the digestive tracts of ruminants. Estimates vary from 49 * 10^9 to 75*10^9 kg C-CH_4, here the figure used is 60*10^9 kg C[1].
-C in manure. This is calculated as follows:
number of cattle[4] * production of manure/number[5] * C content of manure[6].

cow:	1,411 * 10^6 * 3360 * 0.36 * 0.3 =	512 * 10^9 kg C
pig:	840 * 10^6 * 417 * 0.36 * 0.3 =	38 * 10^9 kg C
sheep/goat:	1,649 * 10^3 * 333 * 0.36 * 0.3 =	59 * 10^9 kg C
Total manure		**609 * 10^9 kg C**

Other cattle and fowl etc. are left out of consideration.

[1] Source: World Resources 1990/1991.

[2] Production of grass: only the part that is assumed to end up in cattle. Own assumptions, very rough, are: 5 ton/ha in Europe, 2 ton/ha in the Americas, Asia and the USSR, 0.5 ton/ha in Africa and Oceania.

[3] Source: Intergovernmental Panel on Climate Change 1990; the estimate of 45E09 kg C is from Crutzen (1990).

[4] Source: Eurostat Agricultural Yearbook 1988 (1990), ultimately from FAO.

[5] Data on manure production per animal are based on Dutch figures (CBS, production of animal manure, 1988). An average animal is supposed to be producing 50% of this amount.

[6] Assumption: 0.36 * 0.3, based on formula for organic matter. See 'extraction' with agricultural products.

Human waste emissions:
This factor is treated as a balancing item; according to the inflows and outflows specified above, a number of $99*10^9$ kg C remains for human food consumption. This would mean a yearly consumption of approximately 20 kg C per human or 200 kg of organic matter. The daily consumption would then be 0.55 kg. In the Netherlands, the consumption level is 1.5 kg daily, so the number of 0.55 is within a possible range. Indeed there are estimates of an average daily intake of 0.6 kg/day. It is assumed that the entire $99*10^9$ kg C ends up as a CO_2 type of waste. The distribution over the three waste types is estimated as respiration + faeces (70%) and household waste (30%). Results:

respiration + faeces	$69 * 10^9$ kg C
food based household waste	$30 * 10^9$ kg C
Total human	**$99 * 10^9$ kg C**

3.2 Wood and wood products

Wood is a major environmental sink for carbon, as an accumulation in the environment. The present large-scale deforestation empties this sink at great speed. The deforestation has three main reasons:
-wood cutting for fuel
-wood cutting for economic application in products
-wood burning or cutting for clearing land, without functional use.

Inflow from immobile stocks
The wood extraction for economic use is an extraction from the environment and not from the lithosphere.

Extraction from the environment
To calculate the amount of C involved, several assumptions have been made:
* wood has a weight of 0.7 tonne/cubic metre
* the C content of wood is 73% of the dry weight; same as for cellulose[1]
* wood is 20% water[2].

The world wood production amounts to 3,255,039,000 cubic m [3]. This does not include the non-use cutting. This amount is distributed over:

-wood for fuel:	$1,680.54 * 10^6\ m^3 * 0.7 * 0.73 * 0.8 =$	$682 * 10^9$ kg C
-wood for paper:	$202.44 * 10^6\ m^3 * 0.7 * 0.73 * 0.8 =$	$82 * 10^9$ kg C
-other applications:	$1,372.06 * 10^6\ m^3 * 0.7 * 0.73 * 0.8 =$	$557 * 10^9$ kg C
Total		**$1322 * 10^9$ kg C**

Deforestation for reasons other than functional use is estimated at $1,600*10^9$ kg C[4], a number at the same order of magnitude as all other extractions together.

[1] Source: Chemie Compendium.
[2] My in assumption.
[3] Source: World Resources 1990/1991.
[4] Source: Intergovernmental Panel on Climate Change, 1990.

Emissions to the environment

Wood used for fuel purposes is emitted as a whole. Of the $682*10^9$ kg C involved, 662 is emitted as CO_2, here including all other non-methane hydrocarbons, and the remaining 20 as CH_4[1]. The $1,600*10^9$ kg C from wood burning for other reasons is not distributed over different emission types. A distribution similar to that of fuel use is assumed: 50 as methane, 1,550 as carbon dioxide.

For paper, the assumption is that 50% ends up as waste in the same year, while the other 50% accumulates in the economy in books, reports and other relatively long-lived applications. Paper recycling does not amount to much on a worldwide scale. On the basis of certain data[2] and assumptions[3] the figure of $3*10^9$ kg C is estimated. Based on the same data, a division is made between the several waste treatment types: 75% landfill, 20% incineration, 3% recycling of paper, and 2% "other".

For relatively durable wood applications, it is assumed that all of it accumulates in the economy in the year of production.

There must be a considerable flow from the economic stock to waste treatment from paper and wood products produced in previous years. An estimate has not been made. However, there are some data on the forming of methane on landfill sites. Estimates vary from $20*10^9$ to $70*10^9$ kg C[4], here the average is adopted of $45*10^9$ kg C. Where exactly this C comes from is unknown, but its inflow must be organic, either chemical or biological, and therefore has already been specified.

Accumulation in the economy

Wood	557
Paper	43
Total	**600**

3.3 Miscellaneous

Erosion amounts to $26,000*10^9$ kg soil yearly[5] In this soil there is 1% C[6]. This partly is an extraction from the environment, but partly from immobile stocks as well. Assumed is 50% extracted from immobile stocks and 50% from the environment. Erosion of immobile stocks includes the oxidation of peat by lowering water tables. Erosion brings the material into the biological carbon cycle again. Results:

Erosion, from immobile stocks	130
Erosion, from environment	130
Total through erosion	**260**

[1] Source: Crutzen 1989.

[2] OECD Environmental Data, 1990.

[3] Main assumptions: half of the world's household waste is produced in OECD countries; paper recycling worldwide is assumed to be a factor of 10 lower than in the OECD countries.

[4] Source: Intergovernmental Panel on Climate Change.

[5] Source: WWF Atlas of the Environment.

[6] My own assumption, a rather low figure since poor soils usually erode.

TABLE A.3.2 FLOWS AND ACCUMULATIONS OF CARBON, AS RELATED TO CO_2 AND CH_4, THROUGH THE GLOBAL ECONOMY

Extraction from lithosphere			Deliverance to lithosphere	
	abs.	perc[1].	-Disposal in lithosphere	
-Extraction of:			*CO_2 in natural gas domes	0
*oil	2432	37.5%	*disposal of CO_2 in deep seas	0
*ultra heavy oils, tar, shales	pm	pm	*?	
*natural gas (methane)	1209	18.6%	-Immobilized disposal in environment[6]	
*all coal[2]	2362	36.4%	*final waste in uncontrolled dry sites	?
*peat	2	0.0%	*final waste underwater,	
-Preproduction losses[3] of:			in anaerobic conditions	?
*methane from *methane[4]	45	0.7%		
*coal	35	0.5%		
*oil	14	0.2%		
*other fossil[5]	24	0.4%		
*oil	24	0.4%		
*coal	23	0.3%		
-Erosion	130	2.0%		
Subtotal from lithosphere	*6300*	*97.1%*	*Subtotal to lithosphere*	*0*
Subtotal to lithosphere	0	0.0%		
Net inflow from lithosphere	**6300**	**97.1%**		
Chemical forming in the economy			Chemical destruction in the economy	
-Glowing calcium carbonates			-Forming of carbonates[8]	
*cement production	123	1.9%	*weathering of concrete	12
*plaster production	24	0.3%	-Other stable inorganic compounds	
*CaO in diverse industrial appl.	1	0.0%	*carborundum	?
-Binding sulfur with lime			*?	
*all processes[7]	4	0.0%	-Stable organic compounds[9]	
-Methane from organic wastes	33	0.5%		
Subtotal	*185*	*2.8%*	*Subtotal chemical forming*	*12*
Subtotal chemical forming	12	0.1%		
Net chemical forming	**173**	**2.7%**		

1 As a percentage of total structural inflow, from immobile stocks and by chemical forming.

2 Including lignite.

3 All processes that form a substance and emit it directly are in the scheme twice, once as an inflow and once as an emission to the environment.

4 Including distribution losses.

5 Total emissions of fossil methane are known. They may be calculated from the estimated amounts of current emissions and the ratio of current emissions to fossil emissions. This ratio is reflected in the amount of C_{14} now in the atmosphere. Other sources of fossil methane may be deep ploughing in paddy fields (Lelieveld et al. 1991) and diffuse seeping from permeable layers. See also comments in the discussion to this section.

6 Wastes have been entered as accumulation in the economy. After some time part of these wastes is stabilized and will remain in location for geological periods of time. They should then be transferred to "immobilized disposal in the environment". That quite arbitrary step has not yet been taken here.

7 Primarily flue gas desulfurisation, but also fluidized bed furnaces for coal, and coal gasification.

8 The forming of bones in humans and farm animals constitutes chemical destruction by the forming of carbonates, after extraction from the environment. Burial would effectively remove the bones from the carbon cycle. Incineration and cremation then would be chemical forming and concomitant emission.

9 Plastics may be stable on a geological time scale, if not burned and exposed to light. The chemical forming of plastics for sewer pipes buried underground thus might be treated as chemical destruction. To prevent unclear boundaries between categories, it seems better to place discarded non-burned plastics under accumulation in economy or immobilized disposal in the environment. Chemical destruction would thus be restricted to the forming of stable inorganic compounds.

Total structural inflow	*6485*	*100.0%*
Total structural outflow	12	0.1%
Net structural inflow	**6473**	**99.9%**

Total structural outflow	*12*

Inflow from the environment CO_2 related only, no methane	
-Biotic flows, fixing CO_2	
*wood fuel	682
pulp	82
construction and other	557
*fish and game	11
*grains[1]	174
*grass	505
*other food and fodder	78
*rice roots and stabs in paddy fields	45
-Biotic flows, taking organically bound carbon from the environment[2]	
*deforestation[3]	1600
*erosion of top soils	130
-A-biotic flows	
*CO_2 fixation in concrete	12
* ?	
Subtotal from environment	*3876*

Outflow to the environment related to:		CO_2	CH_4
-Combustion processes			
*fossil fuels		5947	?
*wood		662	20
*wastes	plastics	1	?
	organic	17	?
-Biotic processes			
*from ruminants directly			60
*rice growing in paddy fields			45
*organic materials (landfill)[4]			45
-Chemical forming			
*cement		123	
*plaster		24	
*industrial. applic.		1	
*oxidation of carbon in steel			
-Organic materials to environment			
*biomass[5], faeces + hum.respir		69	?
manure + anim.respir		609	?
*pre-production losses			
oil		24	
coal		23	
*fossil methane spills			118
-Net oxidation of agric. soils			
*erosion[6]		260	?
-Deforestation		1550	50
Subtotal	*CO_2*	*9310*	
to	*CH_4*		*338*
env.	*Total carbon*	*9648*	
Subtotal from environment		3876	
Net emissions	**CO_2**	**5434**	
	CH_4		**338**
	Total carbon	**5772**	

Total inflow into the economy	
*from lithosphere	6300
*chemical forming	185
*imports	none
*from environment	3876
Total inflow	*10361*
Total outflow	9660
Accumulation on balance	**701**

Total outflow out of the economy	
*to lithosphere	0
*chemical destruction	12
*exports	none
*to environment	9648
Total outflow	*9660*

[1] This constitutes net harvest, excluding all plant parts not used commercially, such as roots and stumps.

[2] For the long-term cycle there is not much difference between inflow from CO_2 and organically bound carbon. For the short term the difference is obvious. Living plants directly diminish the amount of CO_2 in the atmosphere, deforestation does not.

[3] Watson et al.(1990) give a survey of estimates of carbon freed by deforestation, including other types of change in land use, ranging from 0.6 to 2.5 GtC per year. The average of the two extreme values has been used here. Similarly, see Houghton (1990).

[4] The carbon part of organic waste handling is treated here as accumulation in the economy.

[5] These flows will partly be as methane, since anaerobic conditions will usually be present to some extent. Solid organic waste has been entered as immobilized disposal in the economy.

[6] Current developments in agriculture are towards lower water tables and deeper plowing. This leads to the freeing of organically bound carbon. If this process goes on uncontrolled, heavy rains and storms will lead to erosion eventually, exposing most organic substances involved to oxidation. Only the latter amounts have been estimated here.

Accumulation in the economy *balance: 701*			
In products[1]			
*plastics		45	
*wood		557	
*paper		43	
*carbon in steel		7	
*other stable carbon compounds		0	
*dump sites	biotic organic	30	
	plastic	5	
	C in steel	1	
Subtotal		*688*	
Rounding errors, mistakes, inconsistent data, = balancing item		**13**	
Total accumulation		**701**	

TABLE A.3.3 COMPARISON OF QUANTITATIVE RESULTS WITH SOME EMISSION ESTIMATES BY IPCC (1990) AND WORLD RESOURCES 1992-93 (1992).

7. Gross emissions to the environment				IPCC data 1985		World Res.data 1987	
related to:		CO_2	CH_4	CO_2	CH_4	CO_2	CH_4
-Combustion processes							
*fossil fuels		5947	?	5100		5811	
*wastes	plastics	1	?				
	organic	17	?				
*wood		662	20		41 [2]		
-Deforestation		1550	50		→↑		
-Biotic processes							
*from ruminants directly			60		56		57
*rice growing in paddy fields			45		82.5		54
*organic materials (landfill)			45		30		34.5
-Chemical forming							
*cement		123		100			
*plaster		24					
*industrial. applic.		1					
*oxidation of carbon in steel		0					
-Organic materials to environment							
*biomass, faeces+hum.respir.		69	?				
	manure+anim.respir.	609	?				
*pre-production losses							
	oil	24					
	coal	23					
*fossil methane spills			118		60		57.1
-Net oxidation of agric. soils							
*erosion		260	?				
Subtotal	*CO_2*	*9310*					
to	*CH_4*		*338*		*270*		*202*
env.	*Total carbon*	*9648*					
Subtotal from environment		3876					
Net emissions	**CO_2**	**5434**					
	CH_4		**338**				
	Total carbon	**5772**					

[1] Including some immobilised disposal.

[2] Methane from wood burning and deforestation together.

4. *Discussion of results*

How do the results of table A.3.2 compare with other sources? Some totals of IPCC and World resources Institute (WRI) have been added for a comparison, see table A.3.3. The divergences occurring are mainly minor, considering the speculative nature of many of the data. One main difference is that on the total emissions of methane. There WRI is one-third lower than our results. The IPCC is halfway between us and WRI. As many possible sources of methane emissions have not yet been quantified (see the question marks in the table below) our estimate still seems rather low.

The analysis of the results in terms of global warming contributions and policy implications is in chapter 5.

5. *Results of the flexible response strategy*

The effects of the flexible response scenario in 2030 have been specified in technical terms and supposed partial quantification in the main text. The results are given here in detail, in the same categories as have been specified for the flows in the year 1987, in table A.3.4.

TABLE A.3.4 EFFECTS OF THE FLEXIBLE RESPONSE STRATEGY OF GLOBAL FLOWS OF CARBON BETWEEN ECONOMY, ENVIRONMENT, AND IMMOBILE STOCKS IN 2030. ALL DATA IN 10^9 KG (MILLIONS OF METRIC TONNES)

1. Inflow into the economy from lithosphere	EJoules	Carb.[1]	4. Outflow from the economy to lithosphere	
-Extraction of:			-Disposal in substrate	
*oil and natural gas	200	4400	*fossil CO_2	3200
*ultra heavy oils, tar, shales, coal and peat	160	3200	-Immobilized disposal in environment	
			*waste biomass	900
-Preproduction losses of:				
*methane from *methane[2]		10		
*coal		10		
*oil		5		
*other fossil[3]		10		
*oil		10		
*coal		10		
-Oxidation of peat by lowering water tables				
*all peat		15		
Subtotal		*7670*	*Subtotal*	*4100*
Net inflow from immobile stocks:		**3570**		
2. Chemical forming in the economy			5. Chemical destruction in the economy	
-Glowing calcium carbonates			-Forming of carbonates	
*cement production		250	*weathering of concrete	25
*plaster production		49	-Other stable inorganic compounds	25
*CaO in diverse industrial appl.		1		
-Binding sulfur with lime				
*all processes		0		
Subtotal		*300*	*Subtotal*	*50*
Net chemical forming		**250**		
Total structural inflow		***7970***	***Total structural outflow***	***4150***
Net structural inflow		***3820***		
Decrease as compared to 1987		***40%***		

[1] Joules are converted to kg C for each source. Dividing by its energy content gives the full mass. Multiplication by its carbon content gives the amount of carbon involved.
10^9 Joule of light oil + methane ÷ 36; * 0.78 = 22 kg C
10^9 Joule of heavy oil + coal + peat ÷ 38; * 0.75 = 20 kg C

[2] Including distribution losses.

[3] This figure has been decreased quite arbitrarily, as no exact sources are now known.

8. Extractions		7. Gross emissions	related to:	
from the environment		to the environment	CO_2	CH_4
-Biotic flows, fixing CO_2		-Combustion processes		
*wood fuel	100	*fossil fuels	4400	?
pulp	200	*wood	97	3
construction and other	1000	*wastes plastics	0	?
*fish and game	15	organic	0	?
*grains[1]	250	-Biotic processes		
*grass	750	*human respiration	0.04	
*other food and fodder	100	*animal respiration	<1	
*rice roots and stabs in paddy fields	20	*from ruminants directly		60
-Biotic flows, taking organically bound		*rice growing in paddy fields		20
carbon from the environment		*organic materials (landfill)		1
*deforestation	500	*net oxidation of peat	15	
*erosion of top soils[2]	100	-Chemical forming *cement	250	
-A-biotic flows		*plaster	49	
*CO_2 fixation in concrete	25	*industrial. applic.	1	
* ?		*oxidation of carbon in steel	0	
		-Organic materials to environment		
		*biomass, sewer	10	?
		manure	100	?
		*pre-production losses		
		oil	5	
		coal	10	
		*fossil methane spills		35
		-Net oxidation of agric. soils		
		*erosion	100	
		-Deforestation	500	
Subtotal	*3060*	*Subtotal* *CO_2*	*5538*	
		CH_4		*119*
		Total carbon	*5657*	
		Net emissions **CO_2**	**2478**	
		CH_4		**119**
		Total carbon	**2597**	
Total Inflow into the economy (TI)		Total Outflow from the economy (TO)		
		CO_2 related carbon		***9688***
		carbon in CH_4		***119***
Total	***11030***	***carbon total***		***9807***

[1] This constitutes net harvest, excluding all plant parts not used commercially, such as roots and stumps.

[2] This flow will partly be from immobile stocks of carbon.

9. Accumulation in the economy (TA)		
	TA = TI - TO =	1223
-In products		
*plastics		80
*wood		700
*paper		80
*carbon in steel		10
*other stable carbon compounds		10
*dump sites	biotic organic[1]	329
	plastic	10
	C in steel	4
Subtotal accumulation		*1223*

[1] Is balancing item.

5.6 CONCLUSIONS ON THE CASES

The aim of the case studies has been a double one. The first is to assess the possibilities for the flexible response strategy and for the macro instruments with which it is built. The primary elements in this assessment are the applicability and the functioning of the individual instruments, and the overall applicability of the strategy. The second aim of case studies are the empirical results themselves. In the application of the macro instruments, the framework for analysis implied in each may shed new light on existing problems. Hence main conclusions on some empirical and normative aspects of the cases will be given as well. The following conclusions first treat the macro instruments individually. Then the flexible response strategy for the cases is assessed. Finally some empirical and normative conclusions are given on each case.

Macro instruments

Extended liability

In neither case can extended liability be expected to contribute much to a solution of the related problems. The applicability is so partial that problem shifting will readily result. Applicability is restricted primarily to local effects of economic processes. With carbon, no application is possible at all. With nitrogen and phosphorus some situational decisions might be effectively influenced. With cadmium, there could be a similar positive effect at some locations. In that case, however, problem-shifting to more hazardous diffuse emissions would counteract this positive effect. On balance, it is difficult to say whether extended liability would have positive effects here. Differentiation of the extension of liability according to substance could perhaps single out the situations where liability might have adverse effects. Different types of liability for different substances would then result with considerable boundary problems. Mixed wastes would be such a very common boundary situation. The apprehension associated with extended liability on theoretical grounds have not been allayed by attractive niches for its practical application. It seems wise not to press hard for extending liability now.

Other structural instruments

In some situations the usual arrangements for property have not yet developed. One such defect, common in most Third World countries, is that any form of visible occupation of a territory may lead to full ownership. Squatting in connection with habitat destruction and marginal and temporary use, may make you rich. Rules on property acquirement as have been usual in Europe at least since the French Revolution could end this destructive mechanism that is also cruel to those who use the forests now without destroying it. Property would be established, private individual, or private common, or collective, with clear entitlements and responsibilities for owners and non-owners. This would not so much be a new macro-instrument as a repair of some defects in the modern way of societal steering.

Standard methodology for life cycle analysis

This standard is not available now. The case on drink packaging has shown that there are forceful incentives for using the methodology. Its availability, including practical tools, would have made the study on milk packaging much easier, cheaper, and better. Its results then would have been more authoritative for marketing. What is the only historical example of the cases has also shown the negative effects of "horizontal politics", with government derived power exercised by some private participants over others. Horizontal anti-waste policy interfered with the functioning of the (proto)life cycle instrument, frustrating its effectiveness in the short term at least.

Other cultural instruments
Substance-specific cultural instruments would not on the whole be effective, if applied at the product level. Some general notions on environmental behaviour of course are useful, as on the suitability of certain types of separate collection. Moreover, producers should take into account the substance information relevant in their situation. This type of information is no longer macro in character. Such information should not necessarily be supplied by governments, especially if macro instruments function broadly.

Substance deposit
The substance deposit has very broad applicability for some substances, but not all. The more durable substances that pass through different processes in society may be brought under the system most advantageously. In some cases the substance deposit might even cover all economic flows of a substance, as with cadmium. Its administrative application seems quite simple, with existing Customs and Excise Offices extending their tasks. The ratio between administrative costs and proceeds seems similar to present forms of excise. The environmental effect thus would be - administratively - created for free. Application at lower administrative levels does not seem possible. The private economic effects caused are based on real costs of emission reduction and on the transfer payment resulting. Real costs, for a given amount of emission reduction, will be the lowest possible in many situations. However, transfer payments by specific industries would give foreign competitors an undue advantage. Broader, supranational, application, e.g. at a European level or of large parts of the OECD, could substantially limit this disadvantage.

Emission taxes
The applicability of emission taxes seems very limited in the diverse cases treated. Where partially applicable the substance deposit would often be applicable as well, but on a broader scale. As an additional instrument, the emission tax is important, to fill the gaps left by the substance deposit. The difficulties in real emission measurement restrict its application also there where gaps have to be filled. These problems in application may be lessened to some extent by technical improvements in emission measurements. For diffuse emissions, such techniques seem quite impossible to develop.

Estimated emission taxes
Solving the problems in emission measurement for taxing may also take another direction, by indirectly assessing the amounts emitted. The NO_x emissions from the use of cars, for example, might be estimated on the basis of actual measurement of emissions in a test run, combined with the actual measurement of the kilometres driven. The application of the estimated emission tax is relevant only where substance deposit and emission tax are both not applicable. Such solutions for practical measurement would have to be developed for all emissions where the inflow and the emission occurs in the same process round, as with NO_x and most CH_4.

Other financial instruments
Financial instruments that do not take processes as their object but products, could not be effectively applied in the substance cases treated. They would lead to a combination of non-taxation and double or even multiple taxation. The incentives for active outflow, as through immobilisation and chemical destruction of the substances would be completely lacking. At a given tax level, their real costs of emission reduction thus would be much higher than those of the other three economic instruments and would be combined with a limited effectiveness

Prohibiting instruments
If relatively macro prohibiting instruments are applicable, the economic instruments are applicable as well. This holds for tradable permits, as noted in Part Two, but also for the general emission design standard and its estimated variant. Their application would generally have the same disadvantages in terms of fixing irrelevant aspects of technology as the product taxes. In none of the cases has it been necessary to use them for building the flexible response strategy, to fill remaining gaps.

Flexible response strategy
The flexible response strategy could be filled in with a negligible role for structural instruments, a limited role for cultural instruments, and a central role for financial instruments. Prohibiting instruments were not required in the cases treated. The quantification of the deposit and tax levels shows that for serious emission reductions high levels are required. For bulk substances, these are on the order of the current price of the substance in the products concerned. For smaller amounts of more hazardous substances a much higher level of deposit and tax is required. The combination of substance deposit and emission tax covers most emissions, except methane. For this substance there is a central role for the estimated emission tax.

Cases
LCA
The limited life cycle analysis executed for milk packaging has resulted in a partial ranking, with polycarbonate taking precedence over carton but not glass, while glass could not be ranked against carton. Only weights on problems could lead to a more complete ranking. If the number of problem categories in the classification increases, a very desirable aim, the chance of single product dominance would become very small indeed. For general use in product development and marketing LCA would have to result in a clear ranking of alternatives, thus requiring an authoritative weighing of problems.

Cadmium
The peculiarities of cadmium relate to its indestructible nature as an element, and its inelastic supply. Therefore, measures at the level of individual products will have hardly any environmental effect, while at the same time causing all real administrative costs associated with application and the behavioral changes induced. Recycling, as a policy instrument, will score badly, with negative environmental effects at real costs. Only inflow reduction and outflow increase may reduce emissions. For inflow reduction, the reduced use of zinc and phosphates is the primary technical option, there to be realized with for example, increased recycling. For the outflow of cadmium, techniques should be defined and developed urgently, given the huge accumulation in the economy that

currently takes place. The substance deposit is fully applicable and has all the desired characteristics.
The level of the deposit might seem high, at several hundred ECU per kilogramme. When compared to the current costs caused by cadmium and other heavy metals in mixed waste processing, however, that deposit would still be low. In mixed waste processing, the costs for emission reduction, now incurred because of ineffective preventive regulation of these heavy metals, are an order of magnitude higher than the level proposed.

Nitrogen and Phosphorus
The emissions resulting from the biological flows of both nitrogen and phosphorus are due to an incredible inefficiency in the application of these substances. Losses of nitrogen in the order of fifty to ninety percent are usual in the agricultural production column. On these flows the substance deposit would be applicable. Very substantial emission reductions would result from a deposit level slightly higher than current market prices. The production in the sector is already too high, with margins too low for long-distance transport. Many of these superfluous and harmful fertilizer factories would have to close.

The non-biological flows of nitrogen, as NO_x, cannot be covered by the deposit system. The emission tax applies to larger installations only. Here there would be a main role for the estimated emission tax.

Energy depletion and global warming
The problems investigated in relation to carbon flows are those of energy depletion and global warming. Energy depletion has been shown in Appendix 1 below to constitute no problem at all. It is the pollution caused by energy use that is a problem. It is therefore the pollution that should be tackled, here the carbon related emissions, not energy use. In the (extended) polluter pays principle, the polluter should pay for his pollution, not for his use of such inputs as energy or steel as such. Other problems caused by the use of fossil energy, not treated here, are the emissions of sulfur, NO_x, and several hydrocarbons. Energy also is an ingredient in still other problems, such as those related to mobility. Even very substantial reductions in energy use of several dozen percentage points would not solve these problems sufficiently. Mobility may keep growing because of increases in the energy efficiency in car design. Reductions per unit of consumption should be much higher and thus would require other instruments, especially in the much desired situation of long-term economic growth.

The one pollution problem treated is that of global warming. CO_2 and CH_4, the two main carbon compounds implied, differ sharply in their contribution to global warming. Methane is more important than CO_2 in the for global warming short run of decades. The data on methane are relatively incomplete so the shares computed are only a lower estimate. Emissions of both substances may be regulated by financial instruments, each requiring a different set. For CO_2 the structural flows related can all be covered by the substance deposit, for methane the emission tax and the estimated emission tax are the prime macro instruments.

PART 6 SUMMARY AND CONCLUSIONS

6.1 MACRO INSTRUMENTS AND MACRO POLICY DESIGN

In the preceding parts and chapters a lengthy design process has been described. Any design process, including that of instruments and policies, has as a central element the making of choices. A pragmatic approach might try to improve on existing instruments and policies. That is not the approach followed here although possibilities for existing instruments have been investigated quite extensively. In the analytic approach taken here, choices are made in a more hierarchical manner, by setting up a framework and filling it in ever more detailed steps. Setting up the framework and specifying it in numerous choices has to begin at quite a high level of abstraction, higher than is usual in stating policy aims. Setting up such a framework is a *von Münchhausen*-like activity; one builds a framework from a multitude of concepts and relations, and then assumes that it stands on its own, firm enough to allow the rest of the building to be added. This hierarchy of choices involves main choices related to the subject of the design activity and to the ways in which it will be approached. The choices involved are not just technical, they relate to how society can and should function. Thus the choices have to be guided by a normative background, described here in terms of principles, as much in line with general principles as now are broadly accepted in Western society. As the building is set up in ever more detail, there are also choices of a more practical nature, on excluding some subjects, on not working out everything to the same level of detail, on not further investigating options that still might be interesting.

What have been the main choices to start with? One major group of choices relates to the macro aspect of instruments and policies to be developed. The design process has been aimed at developing a type of policy instruments that may be applied as comprehensively as possible, at a general level, and that is related to an aggregate analysis of results. Such instruments work through systematically influencing many concrete, individual decisions in society. The set-up is analogue to that of macro-economic policy. There, e.g., the discount rate and the amount of money in circulation together exert a strong influence on "the general level of economic activity", i.e total supply and demand, and total employment. These mechanisms work without most subjects even knowing that they were influenced, let alone that employers had been told to change their production levels and prices and to employ more, or less, persons. Fixing the amount of money in circulation does require very concrete activities, such as spending money by a specific government department on specific items. The macro analysis of the money supply, however, abstracts from these concrete actions. Similarly, the instruments for macro-environmental policy would work at the aggregate level of society as a whole and would be analysed at that level, as contributing to "the general level of environmental quality". Most subjects influenced by the macro instrument would not necessarily know what and how much their contribution to environmental quality is, nor how it is influenced by these macro-instruments. The instrument analysis would not take into account their specific reactions at specific places, only their summed contributions to the aggregate total.

The differences between a micro and macro analysis may be illustrated by the paradoxes that result between them. In economic analysis, at a micro level, individuals would want to save more money, e.g., as a buffer in a politically destabilised period. At the macro-economic level, however, a change in aggregate demand and aggregate income would then result, lowering the incomes of all those wanting to save more to a level where the amount of savings and investments matches again. On average, nobody has increased the amounts saved. If an economic depression then results, savings might easily be lower than they would have been without the change in saving intentions. In environmental analysis, similar paradoxes between micro intentions and macro results may occur. Cadmium applications in batteries could be halted, leading to an improvement at the micro level of this product's effects on the environment. The macro effect on the environment, taking into account the working of the "cadmium system" in the economy, might easily lead to larger instead of lower emissions, through a shift towards other, less easily collectible, applications.

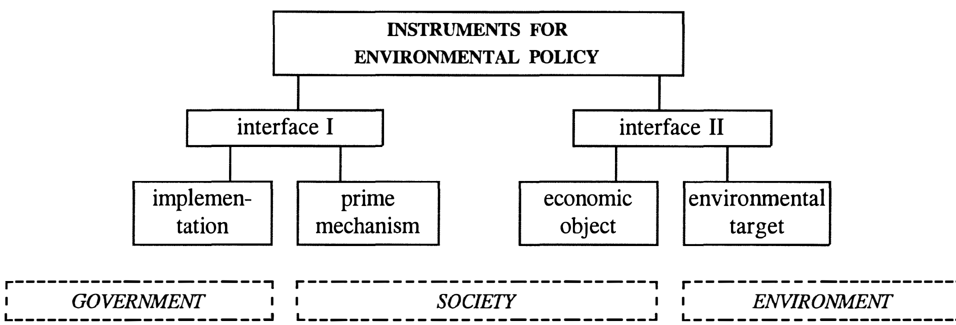
Related to this choice for macro-environmental analysis is the choice on what may constitute the aims at which these macro-environmental policy instruments may be directed. Macro-economics has to restrain itself from the specifics of the sectors, of product quality and consumer satisfaction. Similarly, macro-environmental analysis cannot specify and take into account the specifics of concrete situations as to their "real" environmental effects. The environmental aims can only be specified in terms of overall categories to indicate the general environmental quality, that is in terms analogous to "the level of unemployment" and "the general price level". In that spirit, the general aim of environmental quality has been specified (negatively) in terms of a number of *environmental problems*, such as "ozone depletion", "human toxicity" and "depletion of resources". These more operational aims can only be measured in terms that are independent of specific circumstances and locations, although of course in reality, toxic effects, for example, can occur only at specific locations, regarding specific human subjects and flora and fauna. This restriction is to be brought to mind regularly, to keep in sight what the instruments developed can and cannot do. Just as macro-economic policy cannot force one specific industry to start producing at a specific location, with a certain output and a certain number of people employed, so macro-environmental policy cannot realise a specific environmental quality at a specific location. Other instruments may be required to guide the "distribution" of environmental quality (specified negatively in terms of problems) over different locations, with zoning laws and the appraisal of individual projects (e.g. through EIA, Environmental Impact Assessment), as two central elements. The differing sensitivities of different ecosystems or other elements in the environment, for the different problems they are exposed to, will probably require additional policies at this micro level, even if the general quality were very much improved. Only if problems are fully global, as with global warming and the depletion of a-biotic environmental resources, the related macro policies will generally suffice by themselves. As with macro-economic policy, results will usually obtain through the micro changes brought about.

How will this summarising and concluding part of the study proceed further? First, in chapter 6.2, the result of the design process will be described in some detail, in terms of the framework developed, followed by the specification and choice of the macro instruments and their subsequent combination in the *flexible response strategy,* thus covering Part Two and Three. In that same chapter, the results of the further design and development of the two most interesting and also most macro instruments will be described, the standard methodology for environmental life cycle analysis and the substance deposit, thus covering Part Four.
Next, in chapter 6.3, the central question is how this flexible response strategy can work out in practice, how it can be filled in in terms of the macro instruments to be used, building on the results of the case studies from Part Five. After looking back on these theoretical and more practical results, the next question dealt with, in chapter 6.4, is how broadly this flexible response strategy might be applicable and results might thus be generalised. This final Part ends with a short speculative chapter, 6.5, looking into the political future of the flexible response strategy.

6.2 WHAT HAS BEEN PRODUCED

An analytic framework for the classification and definition of instruments has been developed in Part Two. The essence of this framework is the distinction of three independent subsystems - government, society and the environment - with only two interfaces between them - the government-society interface and the society-environment interface. The third interface, that between government and the environment, has been left out of the model by assuming that all material (in a natural science sense) activities form part of society, while government is engaged in decision-making and in implementing these decisions in a symbolic sense. "Symbolic" here is an encompassing term, including not only normative and empirical information, but also rules, monetary rewards, law, and all other institutional aspects of society. Negatively defined, the symbolic aspects of society are all aspects that are not material in a natural science sense. The causal flow to be analysed in instrument and policy analysis for environmental policy goes one way only, from government, through society, towards the environment. The symbolic activities of government work on the symbolic activities in society. These in turn regulate, and are connected to material aspects of society. It is only these material aspects of society that influence the environment. This conceptual framework forms the basis for instrument definition.

Instruments for environmental policy now may be defined in terms of this very general framework. They have been defined as consisting of two elements, one that creates a government-society interface and one that creates a society-environment interface. Each interface connects at least one variable in the one system to at least one variable in the next. Instruments thus formed are environmental instruments in a strict sense. An example is a tax levied on the owner of an installation, that is interface one, based on the emissions of one substance from that installation to the environment e.g. sulfur dioxide, that is interface two. At each interface a variable from the one system has to be chosen, connected to a variable in the other system. At the first interface, there is an implementational activity towards the subjects involved in society, and a prime working mechanism on these regulatees that is to influence their behaviour in society. The instrument not only has to define the implementation and the working mechanism with its subjects. At the second interface, the material object that forms the basis for regulation also must be specified, as is the environmental aspect of this object in terms of which the regulation operates, that is the target of the instrument. These four elements - implementation activity and working mechanism on regulatee, and object of regulation and target of regulation - fully define an instrument, see the figure below. The special characteristic to note is that the instrument definition is completely independent on the working of further societal mechanisms. Models of the empirical functioning of society, highly debatable and highly absent as general models, may thus be omitted from the instrument discussion, at least at the definitory level.



In the next step, basic classifications of options for each of the four elements have been made. In these, the aim of the study is reflected, i.e. finding macro instruments. Macro instruments can be defined in terms of the framework as having encompassing or aggregated interfaces.

At interface one, it has not been possible to define a "basic unit of implementation". A main variable related to aggregateness in implementation is the administrative level. The specification of the prime working mechanism in society, the second part of the interface, has been based on a further structuring of the model of society, not on modelling society itself. Society, so it is assumed, consists of a limited number of process types as main elements, a structural, a cultural, a financial economic and a material economic one. In the *structure*, the regularities and patterns of society come together, also reflecting the historical investments made in it. The totality of institutions and functioning rules, though not changeable at will in all respects, can be changed deliberately. It is not "given", as a constant. Instruments that change the structure directly, not indirectly through social mechanisms, are thus the first type of working mechanism as part of interface one. The *culture* of society encompasses all knowledge, beliefs and values. A change may be also brought about deliberately here, such a change being the working mechanism of a cultural instrument. Embedded in structure and culture, there is the *economy* of society. It is split analytically into a part where relations are social with a central place for money, and a material part where the physical laws of nature apply. They are the financial economy and the material economy respectively. Changes in the *financial economy* are brought about by financial instruments, that directly change or create prices. Structural and cultural instruments would also influence prices, through changes in demand and supply, but do so only indirectly. Finally, policy instruments may apply at the physical level of the *material economy*, nearly directly, indicating a physical state that should or should not emerge. Material instruments would be the analogous term. Strictly speaking however, it cannot be material instruments as rules and policies cannot apply directly at the physical level. It is always the addressees of the instruments that have to respond. Since at this level regulatees are told by order what not to do and what to do, these instruments have been named prohibiting instruments. The four types of working mechanism may function at different levels of aggregation, with structural mechanisms potentially working most broadly, and in descending order cultural, financial and prohibiting instruments in principle being applicable less broadly.

Interface two specifies the content of the instruments, for environmental policy in terms of environmental targets related to economic objects. The minimal object is one process in society, in its material functioning, and one substance emitted to, or extracted from the environment by this one process. Other types of environmental interference might be taken into account such as sound and disturbances. Such options have not received much attention in this analysis. The challenge here is to specify a more aggregate level than the one-object-one-substance interface that is still feasible as a practical instrument. Objects, as material processes in the economy, can be combined into coherent groups, ultimately encompassing all material economic processes in society. Groups of objects considered are those at one location, those under one owner, those contributing to the functioning of one product, those related to the spending of a budget in a certain way, and - the ultimate aim of macro-environmental policy - all processes in society. Targets, as substances, may be enlarged to groups of substances. The most promising criterion with which groups may be formed is that of environmental problems, such as global warming, ozone layer depletion, acid rain and toxicity. These reduce several substances to one common denominator, through an environmental model with a rather limited scope. Combined, a large number of potential variants of interface two emerges, from which a number has been selected as being "promising" for macro instrument design. First, all substances contributing to all problems, might be related as a target to all processes required for the functioning of one product, the object. The analysis involved here is the life cycle analysis, which specifies the environmental problem contributions of one functional unit of product. Secondly, the analysis of one substance or one group of substances as related to one problem, the target, may be combined with all material processes in society, the object. The analysis involved here is the substance flow analysis.

Combining the two interfaces, a very large number of instruments may be specified. The life cycle analysis may be combined with prohibitions, e.g. forbidding products exceeding some life cycle based product norm. Or products could be taxed according to their life cycle based problem contribution. Or life cycle based product information could be given to consumers who then might change their buying habits. Or final consumers of a product could be made liable for all problems contributed to by the product in its life cycle. Some of these options may or may not be feasible. Taxation as a financial instrument applied to the emissions of one substance from all processes concerned is simply not possible technically. The measurements required are feasible only for some processes, e.g. large stationary sources. Thus only those instruments that are technically feasible are relevant for policy design. If technically feasible, environmental instruments are also to some degree effective. A survey of the literature on policy instruments has been made, focusing on those especially meant for environmental policy. The instruments for environmental policy in the strict sense encountered could easily be moulded into the framework developed, often after their further specification.

Technically feasible instruments, though environmentally effective, might be more or less attractive according to other criteria than environmental ones. In the framework developed, an evaluation of instruments is to guide the further choice on them. Which specific criteria should guide the further development and choice of instruments for macro environmental policy? The range of instruments considered requires quite abstract and fundamental principles on how decision making in society is to be structured. They are the principles of normative political theory. Not all these principles are relevant for environmental policy. Certain elements of equality, for example may be omitted. Equality before the law may be assumed to exist in all types of instruments, at least those considered here, in which case its role as a discriminating criterion stops. Equality of income is another criterion that is better not applied at macro environmental policy. First, there are no indications that environmental policy instruments, though of course related to the incomes of specific groups, are related to the general income distribution. Secondly, should effects occur, these would be small compared to the changes already brought about by instruments specifically aimed at the distribution of income, e.g. cultural instruments such as schooling and financial instruments such as income tax.

The principles selected as relevant here are

Efficiency	Minimal costs of environmental improvements, expressed as environmental improvements per unit of societal costs of the instrument(s)
Equality	Equal treatment of equal assaults on environmental quality in terms of the marginal private costs inflicted by the instrument(s)
Freedom	The lack of impediments on behavioral choices by the policy instrument(s) being applied

The principles may first be applied to individual instruments. There they may further guide the development of instruments. Next, an ordering of instruments according to their attractiveness in terms of these non-environmental principles can be made. The instrument evaluation according to these principles very generally follows the order of aggregation of the interfaces, i.e the "macro-ness" of the instrument. The overall resulting order is as follows.

structural instruments	
* Extended liability	**(?)**
cultural instruments	
* Standard methodology for LCA	**1**
financial instruments	
* Substance deposit	**2**
* Uniform emission tax	**3**
* Estimated emission tax	**4**
prohibiting instruments 1	
* Uniform tradable emission permit	**5**
prohibiting instruments 2	
* General emissions standard	**6**
* Estimated product standard, LCA	**7**
* Estimated emissions standard	**8**

The most macro, structural instrument, extended liability, cannot easily be ranked. Its efficiency in environmental affairs is questionable because of its very partial applicability. For that reason the equality in application will be low as well. On the other hand, it leaves subjects the greatest freedom possible. One defect in the extended liability, through a cultural mechanism, is that, since negligence is no longer a required element of it, the normative meaning of "being liable" is lost. This could bring behaviour now regulated by social norms that are sustained by strict liability into the realm of reasonably "calculable options", in terms of one's own private costs and proceeds. It has been assumed here that the autonomous developments taking place in the Western world will lead to extended liability, but no principle guided choice for it can be reasoned. The prime cultural instrument, the standard methodology for LCA, needs clarification. It is the most macro instrument that could be formulated in relation to the life cycle analysis. The cultural element, an interface factor, is the authoritative declaration of its being standard. The specification of the other interface gives the methodology to be applied in the analysis and the priorities of the problems, to be used for arriving at an overall evaluation in a comparison of different ways to create one functional unit of a product. A possibility to further increase the "macro-ness" of the instrument at the LCA end is to change the functional unit of product into the functional unit of one ECU's worth of consumption, thus comparing the processes required for different ways of spending in their effects on the environment.

No single instrument will ever be able to realise the desired environmental quality, through all required environmental improvements. There is then a number of instruments required that together may bring about the desired results in all the cases that combine to form (macro-)environmental policy. Cases are primarily problem cases but may be broken down into substance cases. How is this mix to be created? The *flexible response strategy* has been formulated as a five step design procedure to create the relevant instrument mix for one case, as a *problem case*. The first two instruments, extended liability and the standard methodology for LCA, which are fully general, work in any case at least to some extent. However, extended liability has a limited applicability and the economy-environment trade-off is weak in the life cycle analysis. Therefore these instruments generally will not suffice. Then case specific instruments have to be applied to the case.

In line with the equality criterion, the procedure is aimed at bringing all decisions in society relevant for the problem case under a regime where the same economy-environment trade-off is created. Instruments have to be added until no further contribution to that aim results, subject to the restriction that each mix is to realise a stated (improvement in) environmental quality. In that sense the strategy is flexible. Generally, the aim of equality can be realised only if just one case specific instrument applies to each relevant decision in society.

The welfare theoretical terms, the policy approach developed here defines a *Bergsonian welfare function*. In such a welfare function there is one part of social welfare that is dependent on the individual utility of citizens, and one part that is defined at a collective level. The environmental problems, with weights attached to them, define the collective term of the welfare function. The individual utility-based part is effectively covered by real private and collective consumption, in market terms. Real costs of policies are in terms of the induced reduction in consumption. With the economy - environment trade-off defined in a consistent manner, the full Bergsonian welfare function results. The two other collective terms relevant in environmental policy, freedom and equality, have not been specified in quantifiable terms. They thus cannot operationally form part of the quantitative welfare function.

In the rather technical Part Four, the two main macro instruments developed here, the standard methodology for LCA and the substance deposit, have been specified in as detailed a manner as is now possible. They build to a large extent on the collective work in the Section Substances and Products at CML.
The first, the cultural instrument of the standard method for LCA still has to be developed further, both the actual analysis part of it and the normative analysis based on the ranking of the different problems a product may help cause. One contribution made here to that development is the method for allocation of multiple processes, at the empirical end. Very generally, processes have several functions that are not all relevant for the one product analysed. An example is the production of chlorine for PVC in one product, that is fully joint with the production of caustic soda and hydrogen used in other products. Another example is the recycling of slaughterhouse waste, with one function in getting rid of the wastes of meat production, and one function in creating other products, such as glues in wall paper production. That allocation method has been specified in principle, by disentangling the social and physical causes of processes. It has not yet been made operational. A main element in the LCA tool that is still lacking is a practically filled in formalised weighting procedure, at the target end. The arguments against such a procedure, related to the fact that LCA can never capture the full truth on the environmental effects of a product, are not convincing. If choices are made anyway one can reverse the reasoning. If they are made consistently, implying certain rationality criteria, there always is at least one set of weights that can reproduce the choices made in a formalised way. Thus general requirements on rationality, such as transitivity and consistency in time, would suffice to make a formalised ranking procedure allowable. There is no doubt that current practical choices, both in politics and in private society, are not made in a consistent manner. The LCA instrument could help create a very broad incentive for environmental improvements, and could create some consistency in environment related decisions in the process.

The second, the deposit system is in principle already operational. It is a financial instrument mechanism applied to the flow analysis of substances. That flow analysis a main structure in which three, not two, systems are distinguished. Apart from the physical part of society (that is the material economy) and the environment there is the substrate or lithosphere to which both are related as their physical surroundings. The three subsystems for flow analysis may also be termed the technosphere, the biosphere and the lithosphere. *Within* each of these subsystems, a substance may be formed, be broken down and accumulate. *Between* the systems there may be flows in both directions. An example are the flows from environment to economy of phosphorus as extractions from the environment, and to the environment, as emissions of phosphorus, together creating net emissions. Environmental problems being man-made problems, are caused by the net emissions to the environment. These nett emissions ultimately result from the difference between structural inflow into the economy, that is extraction from substrate and chemical forming, and structural outflow, that is the flows back to substrate and chemical destruction. Accumulation in the economy will temporarily delay net emissions. Structural inflow into the economy may be mainly from the lithosphere, as with cadmium, phosphorus and carbon, or mainly from net chemical forming in the economy, as with acidifying and nutrificating nitrogen compounds. Structural outflow may be mainly through chemical destruction, as with methane. Back to the lithosphere is an option not yet quantitatively important in the examples. The substance deposit is levied at structural inflow, as extraction from lithosphere and chemical forming, and is refunded at structural outflow, as outflow to the lithosphere and chemical destruction. For geographically defined subsystems of the global system, the deposit is also paid at import and refunded upon export. Effectively, net emissions by society are taxed by this substance deposit system. If applied to several substances in proportion to their contribution to an environmental problem, a "one problem deposit" results. If applied to several problems, there is a "total problem deposit". The relative seriousness of (marginal contributions to) the several problems involved is then implicitly specified in the deposit level for all substances involved.

6.3 APPLICATIONS

The application of the flexible response strategy has a case independent part, i.e. the instruments of extended liability and the standard methodology for LCA, and a case specific part, for all other instruments. No independent picture of extended liability has been sketched. Its influence has been indicated for each case in turn. One indirect working mechanism of the standard methodology of life cycle analysis has been worked out at a case level, the marketing of polycarbonate bottles for application in milk packaging in the Netherlands. All the other cases are substance and problem related, one on toxicity, restricted to the case of cadmium; one on acidification and nutrification, restricted to phosphorus and to the relevant nitrogen compounds; and one case on energy depletion and global warming, restricted to methane and to carbon dioxide related compounds.

LCA on milk packaging

This is the only historical case, based at the time on an LCA method which was not yet worked out or standardised. An analysis has been made in which three main alternatives for the packaging of fresh milk have been compared. The results of that analysis indicated, as effectively as was then reasonably possible, that polycarbonate refillable bottles were to be preferred to the carton/polythene gable top alternative that still dominates the market. The firm commissioning the LCA used the results in marketing the polycarbonate for that bottle. This marketing strategy has so far collapsed, not so much because of limitations in the social working mechanism of LCA, but because of the lack of a standardized methodology on the one hand and interfering horizontal, neo-corporatist environmental policies on the other. In the social process created by the Dutch *Covenant on packaging*, a delay in decision-making of several years has been created already, with unclear prospects for future decision-making. It seems that a consensus on facts has to be brought about in a process where the potential losers to the outcome of the analysis now may have a position giving them a near veto.

Cadmium

The cadmium analysis has been restricted to the level of the European Community. Cadmium gives rise to problems all over the Community. Its diffuse spreading leads to a build-up of concentrations in agricultural soils. Bread made from European cereals is nearing current Dutch health limits, which themselves are already considered too lax. It is a main contaminant of nature areas, surface water and especially underwater soils. Flows of cadmium have been analyzed here using the substance flow analysis. Cadmium enters the economy mainly from the lithosphere, in very low concentrations in zinc ore and phosphate ore, and in much smaller concentrations and quantities in other resource flows such as coal. Its main intentional application is in small scale electricity storage, as in rechargeable batteries. Further applications are in durable pigments, in stabilizers for plastics, in surface treatment, and in several alloys. There is hardly any structural outflow. Cadmium accumulates in the economy, mainly in products, and is emitted. There are considerable differences between countries in the way the cadmium problem is dealt with in environmental policies.

Since the intentional use of cadmium stems from the processing of zinc ore, the supply of cadmium is very inelastic. The ten-fold price increases of the last years have had no detectable influence on the amount of cadmium produced. If cadmium is barred from a particular application, it is expected that, following a decrease in its price, the only effect will be to shift the cadmium to other applications. This aspect has received remarkably little attention, even though it has serious consequences for the manner in which the problems of progressive cadmium pollution should be tackled. The unintentional spreading of cadmium is the most important source of diffuse emissions at present. Measures directed at this specific type of flow will also lead to the shifting of flows, but not to their diminution. If the major source of cadmium in phosphate fertilizer is reduced, the cadmium is shifted to the emissions at phosphate ore processing. If the reductions required for use in fertilizer are high enough, techniques will become economically feasible that extract the cadmium in a concentrated metallic form. It will then find an intentional application. The general conclusion from this analysis is that specific measures that make sense environmentally at a micro level, may or may not be effective at the macro systems level of a country, a region and the world as whole.

Taking this complex state of affairs into account, which strategy, in terms of technical measures, then is the most promising? The best overall strategy is to diminish the inflow of cadmium through a reduction of the extraction of zinc ore and phosphate ore and to direct the flows that still enter the economy towards a safe outflow. Extraction may be reduced by cutting back on the use of zinc and phosphorus and their compounds, by increasing the reuse of these substances, and by using substitute materials. The control of flows, at its best, leads to safe storage of the cadmium for future use, in a concentrated preferably metallic form, or brings the cadmium back to the lithosphere.

Product changes should be assessed primarily by the extent to which the cadmium flow can be controlled ultimately, through collecting, immobilizing and storing the used products or the cadmium contained in them. The level of substitutability of cadmium in the product, now usually the main criterion for whether or not to take measures, is not primarily relevant. In concrete terms, this means that use of cadmium as a stabilizer or pigment in plastic products, largely being an uncontrollable flow, should diminish, even if near *substitutes* are *not readily available*. To compensate, the concentrated use in products that can readily be collected should increase, *even if substitutes are readily available*. Storage cells and batteries, for which there is a rapidly growing market, are a typical example. This application should continue, provided it is coupled with effective collection and followed by permanent immobilisation as in the lithosphere. If cadmium-free storage cells and batteries should appear on the market as substitutes for the nickel-cadmium versions, this useful outlet for cadmium would disappear. From the environmental viewpoint this makes research into cadmium-free batteries unwanted, at least as long as cadmium-containing ores are processed and there is no other recoverable product to take up the cadmium currently used in batteries in a similarly concentrated way.

The recovery and reuse of cadmium after collection leads to a more efficient use of cadmium as a basic material. Given the inelastic inflow of cadmium, that means a broader application but not a smaller inflow or lower emissions. Ultimately an equilibrium situation arises in which the cadmium inflow in phosphate and zinc ores still finds its ways out of the economy, partly as emissions into the environment, increasing with the percentage recycled. Collecting and recycling Nicad batteries as contemplated and partly effectuated in most Western countries, will lead to increased diffuse emissions as compared to collection followed by safe storage and outflow back to the lithosphere. Recycling will even worsen the current situation. This results from the fact that all applications, including those that are non-controllable, will then increase with increased reuse. Immobilisation of collected cadmium is in the end the only effective measure, and easiest with concentrated applications. Recycling is thus *not* to be pursued with collected cadmium, contrary to the aim of the current EC directive, see Council of the European Communities (1990c). This prescribes that cadmium containing batteries receive a "recycle" sign and that measures to promote recycling be taken.

Structural outflow, preventing future emissions, requires some form of storage. Storage can take place in such a way that subsequent reuse is possible if then desired, i.e. strategic reserve. It might be done with metallic cadmium directly purchased on the market, before any useful application. If depletion is not a threat, as seems most probable, the best solution for cadmium is to bring it back to the lithosphere.

How can the flexible response strategy for cadmium be developed? Which policy instruments could implement the controls on cadmium required?
Liability cannot play a significant and independent role. It might prevent local emissions but in the process would cause more diffuse emissions that are also more harmful. The life cycle analysis, not taking into account indirect economic effects of the kinds described in SFA, here gives an irrelevant signal on cadmium. Information on only cadmium which is in products is not an effective information approach. The complex relations between different decisions make it difficult to supply the right information.

A deposit on all cadmium flows is the preferred financial policy instrument in the flexible response strategy for cadmium, and the only one required for equal application to all cadmium related decisions. The deposit would be paid on all imports and all extraction from the environment, with a refund upon export and immobilised disposal in the economy for future use or upon permanent disposal in the lithosphere. A deposit amount of 185 ECU/kg cadmium (no emission taxes are required) will be high enough to tackle the main flow - cadmium in batteries - while various other flows will also be influenced. The full effects of the deposit have not been quantified in this study, but long-term emissions will be reduced to a fraction of their current levels. If the deposit is applied to several heavy metals, the costs of the processing of then mainly clean flows of household sewage and household waste, now soaring in most Western countries, could drop to a fraction of those levels. If no useful applications of cadmium remained, the full costs of cadmium storage would be borne by the users of zinc and phosphate.

Nitrogen and phosphorus
Phosphorus and the nitrogen compounds singled out for analysis both contribute to overnutrification. The nitrogen compounds involved also cause a substantial part of acidification, both directly in the form of nitric and nitrous acid, and indirectly through denitrification of emissions to air of NH_x. These acid emissions also contribute to nutrification. Some flows, like those going directly to soil and surface water, contribute to nutrification only. The structural inflow of nitrogen is nearly exclusively through chemical forming, intentional in fertiliser production and unintentional in most combustion processes. The flows of nitrogen oxide now are completely separate from those of the other nitrogen compounds, being directly emitted after chemical forming. The structural inflow of phosphorus is exclusively through the extraction from the lithosphere. The structural outflow of both substances is limited, the inflow mainly leads to net emissions, with some chemical destruction in the case of nitrogen and some storage in the economy in the case of phosphorus.

Technical measures that limit net emissions are those that ultimately reduce structural inflow and increase structural outflow. For the intentional uses of nitrogen, the main solution is in the increased efficiency of fertiliser, in the recycling of manure, and in the prevention, chemical destruction or reuse of ammonia that may be formed from manure. For non-intentional inflow at combustion processes, any other processes might be used or the nitrogen compounds formed may be denitrified again. In the technical solutions, the flows of nitrogen oxide, as from power plants, might be combined with ammonia, as from manure processing, together either forming N_2 and water, or fertiliser, depending on the circumstances. For phosphorus, there only is intentional use. Increased efficiency in the cycle, by better techniques of manure processing and application, and by other techniques of sewage processing, are the main options through reduced inflow, thus also reducing the cadmium inflow. Immobilised storage is the only outflow-increasing option.

Extended liability will hardly be applicable to these very common flows. Only in exceptional cases, as with specific nature reserves threatened or with ground water used for drinking water, might there be some effects. The general effects of the life cycle methodology will remain limited. The case-specific instruments here are more diverse than in the previous case. The substance deposit is applicable to the main intentional uses of nitrogen and for all intentional flows of phosphorus. For the non-intentional uses of nitrogen, which are mainly in combustion processes, the emission tax is applicable only to larger stationary sources. The estimated emission tax is applicable to the remaining smaller and mobile installations. The level of the deposit and the tax suggested has been indicated by the costs of manure processing. At that level, of ECU 1.20 per kilogramme nitrogen or phosphorus, a substantial reduction in NO_x and ammonia emissions will occur, thus contributing to a reduction in acid emissions. The effects on over-nutrificating emissions, that also are to soil and water, will be more limited. The overall level of reduction of N emissions, after ten years, has been estimated to be in the order of 30%.

Energy and global warming

In the discussions on energy taxes, there is a mixture of arguments related to energy depletion and to global warming. The energy depletion argument has been investigated. The results of that analysis indicate that there is no reason to fear depletion. A conservative estimate of ultimately extractable fossil fuels indicates reserves of several thousands times current energy use per year. It is based on the assumption that the current exponential growth in proven reserves will not only flatten out to linear growth, but will come to a full stop in a few decades and that some of the non-conventional reserves will become extractable in due time. If the breeder reactor becomes operational, and there is several centuries of space for experiments based on fossil reserves, there is an amount of energy in the form of uranium and thorium available that is at least as large as the total fossil reserves. Flow energy, especially solar energy generating electricity and hydrogen, may quite easily cover total energy demand, at a surface coverage of only 10% of all tropical deserts, with a transforming efficiency as is already common now. Biotic energy, as a single main basis for energy supply, seems a less attractive option. However, it may usefully other forms. For a long time to come, the choice of energy source seems to be based on costs, not on any real shortages. It seems a waste of the limited regulative capacity for environmental policy to apply it to this energy depletion question. No policy analysis for energy depletion has been developed. Quite opposed to energy depletion, the problem of global warming, though variously assessed in its effects, causes such severe risks that policy development seems a short-term must.

The analysis of global warming has been restricted to carbon flows, in two forms. One is that related to all carbon dioxide forming compounds, the other is that of methane. The structural inflow of both compounds primarily is through extraction from the substrate. The inflow of methane is also through chemical forming, in agriculture by ruminants and paddy rice growing, and also in other processes leading to the fermentation of organic materials. Quite important flows of methane emissions also originate from coal mining and from leaks at the transport of methane. There is hardly any structural outflow of the carbon dioxide part of the flows. Methane of course is mainly burnt, thus contributing to carbon dioxide emissions. In this sense the methane analysis is superimposed on the more encompassing carbon dioxide analysis.

The results of the analysis of the two flows are quite striking when expressed in terms of their global warming contribution. The net emissions of carbon dioxide do not contribute much more to global warming than those of methane, using the latter's one hundred year base global warming potential, $GWP_{100} = 21$, for conversion. If the climate forcing effects of the coming decades are deemed important, assuming, for example, that higher emission reductions will become possible by then, the conversion might be done on the twenty years base, as $GWP_{20} = 63$. Then methane makes a substantially more powerful contribution to climate change than carbon dioxide. The amount of methane emitted into the atmosphere is halved in less than a decade, that of carbon dioxide in well over a century. In that short period, the effects of methane, per kilogramme, are twenty times higher than those of carbon dioxide during a century. These results are striking especially when compared to those of the IPCC survey that have become the accepted basis for policy development. There, the methane contribution is given not as a percentage of net emissions but, incorrectly, as a fraction of gross emissions. Then its preponderant contribution to the climate effect is lost in the flows to and from the environment of organic growth and breakdown, which is carbon only dioxide related. In the IPCC data, carbon dioxide takes a share of sixty-one percent of emissions and methane only fifteen in total GWP contributed by all emissions in 1990. The structural contribution of current emissions of methane is as important as that of current carbon dioxide emissions. It thus deserves at least the same degree of policy attention.

The techniques of emission reduction differ quite substantially between the methane flows and the general carbon flows. The main source for methane is associated with coal mining. These losses may be lowered, e.g., by oxidising the ventilation air or by extracting the methane before mining operations start. Gas leaks in transport, a second main source, may be diminished by fewer leaks, flaring and recompression. Structural contributions by chemical forming all are based on anaerobic fermentation, especially in organic wastes, in paddy fields and in ruminants. Other techniques for waste processing and storage are available. Also for paddy rice growing alternatives are available, though often not for the farmers now involved. The intestinal functioning of ruminants seems scarcely subject to change at present, but who knows. Volume reductions in ruminants would be the main technique there for the time being.

The techniques for emission reduction in carbon dioxide related flows are different. They relate much more clearly to reduced structural inflow and increased structural outflow as independent flows. For reducing structural inflow, higher energy transformation efficiencies and shifts to low carbon fuels are the main options available and widely researched, as are shifts to non-carbon based sources of energy and reductions in final energy use. Direct solar power is the main long-term option here, as soon as its costs fall below that of fossil energy costs, probably also decreasing. At the structural outflow end, research is starting, with two promising options. One is underground storage, in old gas fields and other permeable underground reservoirs and in the application in oil fields for enhanced oil extraction. The other is enclosing them in an ice crystal lattice under very high pressure, such as clathrate structures. The pressures required indicate such storage at the beds of the ocean, deeper than 500 meter. Quite related technically would be the pumping the CO_2 into the deeper layers of the ocean where it would take thousands of years for surfacing. One other option is to store the carbon containing organical materials (e.g. wood) permanently, either in dry or in anaerobic (non-fermenting) form. That storage option is a low-tech method feasible for many waste flows. The immobilised storage of organic wastes would lead to the structural outflow of a non-structural inflow, from the environment. At balance, however, it makes no difference where the structural outflow originated. Burning wastes, with a usually low efficiency in energy recovery or none at all, is one of the worst options in waste processing as far as carbon dioxide emissions are concerned.

With carbon dioxide and methane, extended liability and life cycle analysis do not contribute much to emission reduction, as compared to the high reduction ratios that are supposed to be needed. Thus, case specific instruments are required. Most carbon dioxide related flows can be brought under the substance deposit scheme. At the structural inflow end, there is not much difference with a carbon tax on fossil fuels, apart from minor flows as with cement production. The quite essential differences are at the structural outflow end. Storage is a serious option there that would never emerge with the emission tax. Criteria for immobilised storage would have to be specified, especially for waste storage. The emission tax resulting from the deposit system would be much more acceptable in international trade than a product tax. This would be even more the case since emissions other than just those of originating from fossil energy carbon would be taxed as well. That is exactly what the next part of the flexible response strategy is about. All flows not yet covered by the substance deposit would be taxed, either through direct measurement of the emissions or through their estimation. This is the case with calcium carbonates used in cement production and flue gas desulfurisation and especially with methane. With methane, the application of the substance deposit is nearly absent while the possibilities for emission measurement now often will be limited as well. Measurement techniques might be improved substantially, either by sophisticated measurement methods or with the help of modelled measurement, with actual measurement at some distance. The level of the carbon tax would have to cover the costs of permanent storage. The most precise cost predictions, by Shell for the Dutch Ministry of the Environment, indicate costs of ECU 0.075 per kilogramme carbon stored in gas fields, originating form an coal gasifier based electricity plant using pure oxygen for burning. The emission reductions in terms of global warming have been estimated quite crudely. However, as methane emissions and the structural outflow of carbon have been included, these more realistic estimates of reductions possible are larger than those indicated in the other studies mentioned.

When comparing the flexible response strategy for global warming with the energy tax proposals that have been discussed in the political arena, the level chosen seems modest. It is equivalent to a deposit payment on oil of slightly under \$11.00 per barrel. The proposal of the European Commission amounted to \$10.00 per barrel, the US Congress investigated a level of carbon taxes equivalent to \$13.00 per barrel, and the Wolfson

Commission in the Netherlands investigated a level of $40.00 per barrel. Neither of the policy studies on climate related energy taxes investigated the possibilities for increased structural outflow, only volume reductions at inflow were modelled! Neither policy proposal paid attention to the emissions of methane, that are as important for global warming as those of CO_2.

The flexible response strategy: applicability in the cases

The structural, cultural and financial instruments that have been developed suffice to make operational the flexible response strategy. Extended liability can hardly play a role for the cases investigated. The role for LCA is limited per substance and problem but very broad over all problems and over many types of decisions. Apart from environmental effectiveness and the high score on principles, it also is culturally attractive in that it gives insight into the personal involvement in environmental problems.

The policy development cases, all related to problem - substance combinations, indicate that the financial instruments of substance deposit, emission tax and estimated emission tax together can cover virtually all the flows concerned in an equal manner. If applied in a context of reasonably functioning markets, not necessarily all fully competitive, their combined influence creates a trade-off in society in all decisions concerned so that the social costs incurred per unit of environmental improvement are high and equal, leading to large environmental improvements in a dynamically efficient manner. "Equality-before-the-policy", in terms of private costs per unit of environmental improvement through some decision, seems most readily possible through the combined covering application of financial instruments alone. Adding prohibiting instruments, even if assuming that knowledge on real private costs induced by them is available, would lead to a difference between decision-makers in society in the transfers they pay to government, thus making an equal trade-off impossible. Freedom left to regulatees also is as high as is compatible with realising a satisfactory level of environmental quality. This is a very reassuring result: there really exists an attractive macro alternative for the now dominant prohibiting regulations at the micro or at best meso level.

The financial instruments chosen in the flexible response strategy by nature also have a taxing function, with proceeds for governments. At a given level of government spending and deficit, these environmental taxes therefore replace other taxes. Thus, they might be called *ecotaxes.* The central role for financial instruments in the flexible response strategy does not mean that it is similar to the ecotaxes now proposed by the environmental movements in Northwestern Europe. These ecotaxes are to a large extent product taxes, very different from the process oriented substance deposits and emission taxes proposed here. Products have only an indirect relation to processes and their emissions. Ecotaxes as product taxes thus can only be defined as financial instruments in terms of their working mechanism. They cannot be defined in terms of a clear society-environment interface and thus are not instruments for environmental policy in the strict sense defined here. The superiority of instruments with an explicitly defined society-environment interface is most vividly illustrated by the commercial chances of some environmentally attractive process techniques. One technique for combined coal gasification and electricity production is an example. This technique is capable of astonishing reductions in nearly all emissions associated with fossil energy use. Sulfur emissions, NO_x emissions and heavy metal emissions may stop altogether, with the additional possibility to fully destroy organic chemical wastes. The concentrated CO_2 coming out of the process can be stored in the lithosphere. *Energy taxes and other input taxes or product taxes can never make this technique profitable.* Substance deposits and equivalent emission taxes on carbon, NO_x, cadmium and also sulfur, at the levels indicated, would make this cleanest of all techniques for gas and electricity production profitable tomorrow.

6.4 BROADER APPLICABILITY

It seems that the substance deposit is most suitable for substances that pass through several economic processes before eventually being emitted. This is the case with all substances that are extracted from the lithosphere. Their net emission reduction requires either reduced inflow from the lithosphere, where deposit payment will usually be feasible, or increased outflow to the lithosphere, where deposit refunding will not pose severe administrative problems either. All substances that may be defined as problem substances at the level of elements can be covered this way. Main examples are sulfur, phosphorus, and heavy metals such as cadmium, mercury and lead. Also compounds, as groups of substances, may be treated in this manner. With these, structural inflow and outflow by chemical forming and destruction is also possible, sometimes perhaps taking place on a small scale that might make refunding difficult. Such difficulties, however, have not been encountered in the major cases investigated.

Also substances entering the economy through chemical forming, if they pass through several processes before eventually being emitted, may best be brought under the deposit system. A main example treated is intentionally produced nitrogen compounds. HCFCs, and the CFCs not yet phased out, would be likely candidates, with deposit payment at their production and refunding at their destruction.

If substances enter the economy and leave it in the same process, the deposit system if applied would effectively be equal to an emission tax at that process level, NO_x formed in combustion processes being a main example. It then is better to designate it as such and base payment on actual emission measurement where possible. Thus, for such substances, the emission tax may be applicable rather than the deposit system. Its application seems restricted to larger installations where continuous emission measurement is possible at reasonable costs. The limitations on actual emission measurement may be reduced substantially if measurement techniques to that end were to develop further. There do not seem to be fundamental technical reasons why the emissions of NO_x and other substances from cars could not be metered and paid for at regular intervals. For all non-diffuse emissions actual measurement might become a viable technical option. For the hazardous emissions still remaining, the estimated emission tax seems applicable. If the estimate comes near to real measurement and the costs of estimation are not too high, it is an option that is certainly preferable to prohibiting instruments. With high costs of very indirect measurement the advantages of the estimated emission tax over prohibitions disappear.

One main problem in broadly setting up the flexible response strategy is the way all compounds are to be grouped. If one would go for precision, any individual substance would have to be regulated individually. This is not really possible, at least not with financial and prohibiting instruments, given the very large number of substances now produced. Grouping is necessary. With heavy metals, a choice has been proposed to treat all compounds in relation to the amount of the heavy metal only, even if different compounds differ drastically in their effects. Metallic mercury is hardly toxic, while methyl mercury is extremely poisonous. The reason for treating them together is first the ready transformation of one compound into the other, after emission, and, secondly, the ease of application of the instrument. If all compounds could be treated at the relevant element level, their regulation would be very easy indeed, though at the price of imprecision. It could be an option for the very large number of often very hazardous halogenated organic compounds to regulate them with a "chlorine deposit", a "bromide deposit", etc. Some substances with extreme hazards, such as dioxin, could then be regulated separately, on the one extreme, while some extremely non-hazardous substances, e.g. chlorine bound in a polymer such as PVC, would be exempted. Then the emissions from burning the latter not yet regulated substance, in terms of acids, PCBs, and dioxin, would be taxed at the incinerator, with all other substances being formed there. A relevant grouping of substances has not been proposed in this study. The analysis here suggests, however, that main environmental problems, excluding purely local ones and those of extremely toxic compounds, can be brought under the flexible response strategy at the level of financial instruments.

6.5 PROSPECTS FOR MACRO-ENVIRONMENTAL POLICY

The two most macro instruments, extended liability and the standard methodology for life cycle analysis, are developing. Extended liability, a mixed blessing, seems to result from a broad cultural change in society, without a fierce public discussion between those for and against it. The life cycle analysis is being developed both at a private level and at a public level. As a private tool, for firms, its level of specification will remain limited to a main outline, at the level of a code of practice or as a standard specified by ISO, and similar organisations at the national level. The impetus for the full specification of a standard methodology can come from the public sphere only. Governments active in the field, mainly in North America and in Northwestern Europe would have to increase their efforts towards both the formation of general theory and the creation of tools. The allocation problem in the inventory component, the choice of the environmental problems to include in the classification component and how to include it operationally, and the weighing procedure in the evaluative component of the life cycle analysis are central elements to be developed further in a public context. The practical tools for application, software and a data base on general processes, may be a public responsibility also, at least in the early stages of application of the standard method. The life cycle assessments made with the standard analytic tool and the practically supportive tools may remain a private affair. It would seem that a more dedicated decision in favour of these goals is necessary, with intergovernmental coordination becoming necessary in due time. Prospects seem favourable. The environment - economy trade-off implied in the instrument is low, which means that costs per unit of environmental improvements induced will be low, compared to that of the financial instruments specified in the cases. Its influence will be broad and dynamic. For each specific decision it will remain limited, however, due to the only low environment - economy trade-off that will emerge voluntarily. When higher costs are required for the extensive environmental improvements that seem necessary, other instruments will have to be added.

For the additional development of more effective policies, in problem cases, the introduction is required of the three financial instruments in the strategy, the substance deposit, the emission tax and the estimated emission tax. Conceptually, these instruments are ready. Contrary to extended liability and the methodology for LCA, their introduction depends on a political struggle between those who see the general advantages and those who will feel the specific pressures of their introduction, by having to pay for their emissions, even indirectly. There is one right way only to ease that pressure somewhat, that is by starting the instruments at a low level of deposit and emission tax, and then increasing the level gradually and predictably to the desired value. This approach has been adopted in the energy tax proposals of both the European Commission and the United States government. The public finance advantages of introducing environmental taxes, in terms of a reduction in other taxes, will usually be spread in the population, with a net transfer paid by those being responsible for the highest emissions. By introducing financial instruments for several substances at the same time, these transfers to-and-fro will tend to even out.

At the time this final chapter was being written, the first-ever moves towards broadly applied financial instruments in macro-environmental policy, the introduction of carbon taxes on fossil carbon ("from lithosphere") seemed to have been halted. Their introduction in the EC has been made dependent on a broader international introduction, especially in the US and in Japan. In the US, the proposals of the Clinton administration[1] on a substantial carbon tax have been grounded, by special interest politics in the Senate, and have broken down into a number of excises on different fuels, with many exemptions (as of 1-6-1993). Effectively the newer proposals now move into a still worthwhile direction that, however, had been created in Europe and Japan a long time ago already. Chances on a European introduction now have thus become very slim. What seems required for a broad introduction of the flexible response strategy is a combination of daring policy choice with broad international coordination, since the disadvantages of going alone are very considerable with financial instruments. Such a combination is highly improbable in normal times. If ready administratively as a broad package, it seems that its introduction could take place on a wave of environmental concern as could be created through social mechanisms or through a major environmental disaster.

[1] Developed attunedly with the proposals of the EC, Delbeke, DG XI, personal communication.

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