Procedures for the Establishment of Standards

Final Report
Volume 2

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PREFACE

This final report summarizes two years of research on analyzing procedures for the establishment of standards. The research was sponsored by the Volkswagenwerk Foundation and jointly carried out at the International Institute for Applied Systems Analysis at Laxenburg and the Kernforschungszentrum Karlsruhe. The final report is meant to be both a problem-oriented review of related work in the area of environmental standard setting and an executive summary of the main research done during the contract period. The following eleven technical papers (Volume II of the Final Report) are reference reports written to accompany Volume I. They describe the studies and findings performed under the contract in more detail, and they have been either published as IIASA Research Memoranda or as outside publications, or were especially written for this report. These technical reports are structured in four parts:

- policy analyses of standard setting procedures;
- decision and game theoretic models for standard setting;
- applications of decision game theoretic models to specific standard setting problems;
- biological basis for standard setting.

TABLE OF CONTENTS

I.

POLICY ANALYSES AND STANDARD SETTING PROBLEMS

ENVIRONMENTAL STANDARD-SETTING: EFFICIENCY, EQUITY, AND PROCEDURAL PROBLEMS*

Giandomenico Majone

Technical Report No. 1 written for the Volkswagenwerk Foundation Project, "Procedures for the Establishment of Standards"

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Environmental Standard-Setting: Efficiency, Equity, and Procedural Problems

Giandomenico Majone
Russell Sage Foundation

Even more than executive agencies, regulatory bodies today find themselves in the eye of a storm of criticism and distrust. Two main approaches have emerged as possible solutions of what many people see as a general failure of governmental regulation. The first alternative, strongly advocated by economists, is deregulation. The second alternative, favored especially by the courts, relies on institutional and procedural innovations designed to permit all affected interests access to the process of regulatory agency decisions, and to improve the quality of agency deliberations.[1]

The capture of the regulatory commissions by the very interests they are supposed to control, and the ultimate futility of economic regulation, have been repeatedly discussed, and to some extent documented. These facts have been interpreted as providing additional confirmation of the virtues of the market, and of the need to reduce governmental intervention. This critical literature has its counterpart in many analyses of the environmental problems, where the administrative approach to pollution control (prohibitions, standards, incentives, and so on) has been severely criticized for its lack of effectiveness, and for its tendency to become "a political process entailing bargaining between parties of unequal power."[2] Effluent charges and related market-oriented techniques have been proposed as alternative approaches that, by their automatism and in conjunction with the integrated management of river basins and airsheds, "would reduce the scope for administrative discretion and bargaining."[3] But these normative conclusions overlook one important point: the same forces that influence and distort the regulatory framework, will also affect other approaches, by the same or by different methods. The comparison between, say, an uncorrupted system of effluent charges, and a regulatory machinery captured by special interests, is a specious one. Where effluent charges have been used, for instance, in France, they have proved to be as subject to bargaining and as conditioned by considerations of political and administrative expediency, as standards, licenses, and other regulatory measures. Thus, the search for a system that "would resolve most of the political conflict over the environment in a highly visible way," is bound to lead to disappointments and, in the final analysis, to doubts "as to the ability of a pluralist political system to make wise choices in issues of this sort."[4]

What are wise environmental choices? Traditionally, efficiency and equity have been used as the main evaluative criteria, but the intense controversies surrounding many regulatory decisions suggests that legitimacy -- the generalized disposition to accept, within limits, decisions independently of their substantive contents -- is equally important. It follows that it is not sufficient to examine the <u>substance</u> of a proposed policy in terms of economic and distributional impacts; the <u>procedures</u> by which decisions are reached must also be carefully considered, for in the absence of generally accepted criteria of "truth" or "rightness," legitimacy depends on procedures. Present concern with questions of legitimacy in the environmental field is revealed by the fact that regulatory agencies in the United States, Germany, and other countries, have come under increasing pressures from the courts to formalize their decisionmaking process by such procedural requirements as hearings with sworn testimony and rights of cross-examination. Such developments are bound to have far-reaching consequences for policymaking as well as for policy analysis.

This paper reviews the current state of the debate on the relative merits of the market and administrative approaches to pollution control, and analyzes some recently proposed institutional mechanisms to improve the quality of environmental decisionmaking. Its main conclusions are twofold: first, in spite of their alleged superiority, pollution charges and similar price-like devices cannot replace standards as the main tools of environmental policy; second, the most urgent problem facing policymaking and analysts today is that of designing procedures capable of increasing the rationality as well as the legitimacy of the standard-setting process.

Market Approaches

Under this heading I shall consider effluent charges, pollution rights, and a number of proposals to internalize environmental externalities through utilization of the existing legal system, or a better definition and enforcement of property rights.

Consider the case of a chemical factory producing product P and discharging its wastes into a nearby river, or polluting the air with its fumes. These discharges reduce the quality of the environment and its suitability for a number of alternative uses. Since water and air are common property, their services (in this case the service of carrying off wastes) are not sold. The costs in terms of reducing environmental quality is thus overlooked by the price system, and failure to account for such costs leads to an oversupply of P and an undersupply of the benefits which are reduced by pollution. This is the efficiency problem. If the damage cost of pollution were internalized, resource use would become more efficient: the price of P would be higher, less P would be produced, pollution would be abated.

To consider the simplest case, suppose that both the level of discharge and the cost of damage are monotonically increasing functions of the level of output P; and that, in the short run at least, nothing can be done to reduce the damage done per unit of output of P. The situation depicted in Figure 1 results.

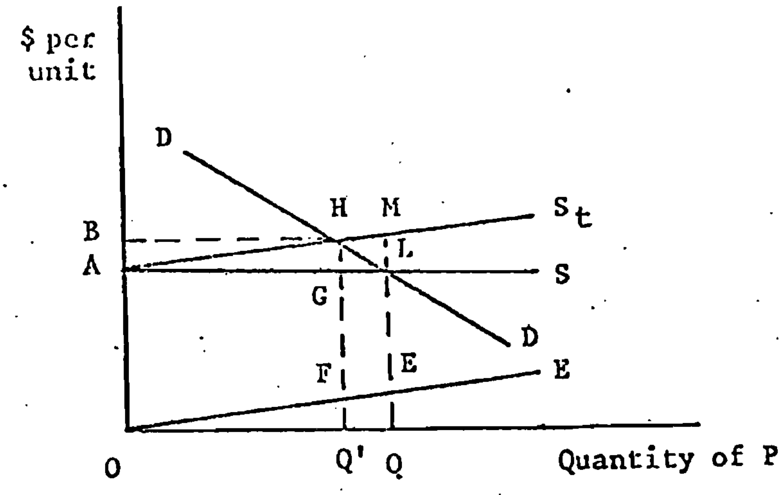

Figure 1

Here AS is the firm's supply schedule (under competition), OE the marginal damage cost (e.g., the loss of water quality valued in dollars per additional unit of P), and AS_t, the sum of the costs of production and the environmental costs. If DD is the demand schedule, a quantity OQ of P would be produced in the absence of government intervention. On the other hand, efficient output (taking into consideration also the environmental costs) should equal OQ'. To achieve such an efficient output without direct regulation, it is in principle sufficient to impose a tax or charge equal to AB. Assuming compliance, output is reduced from OQ to OQ', and pollution damage is reduced by the amount of Q'FEQ.

This simple case is sufficient to bring out the basic logic of the pollution (or effluent) charges approach. The more realistic case, in which technologies are no longer assumed to be fixed, and the pollution level can be reduced by changing the methods of production, presents no major analytic difficulties; at least as long as marginal damage costs and marginal abatement costs are supposed to be known. The public authorities simply set a charge or price equal to the marginal damage for each unit of waste. Polluters would then decrease their waste flows as long as the marginal cost of doing so was less than the price for discharging, settling at the optimum where marginal treatment costs equal the charge.

But even at this level of abstraction, one practical complication arises. Namely, as pointed out by Baumol and Oates,[5] the optimal charge level is not equal to the marginal net damage caused <u>initially</u> by the pollution-generating activity, but rather to the damage it would cause if the level of the activity had been adjusted to its optimal level. Thus, if a factory currently causes 50 cents of damage for each additional unit of output, but suitable changes in the technology could cut the damage to 20 cents, the optimal charge should be set at 20 cents per unit of output. A unit charge of 50 cents would lead to an inefficient situation, where pollution is reduced beyond the range over which the marginal benefit of decreasing the externality exceeds its marginal cost. Thus, while it is difficult enough to estimate the environmental damage currently produced, a rigorous application of the charges approach actually requires that we estimate the damage that would result in an hypothetical world where technology, production methods and institutions have been optimally adapted to prevalent environmental conditions.

Under an alternative market approach, the public authority does not attempt to tax polluters at a predetermined rate. Instead, after setting a limit on the allowable discharge in the region, the authority auctions off pollution rights to polluters willing to bid the highest price for them.[6] Under static conditions, the outcome prevailing under a system of pollution rights can be made identical to the one prevailing under the charge system. Consider, by way of illustration, a situation with only two polluters, X and Y, each discharging the equivalent of 100,000 pounds of FSUOD a day (First Stage Ultimate Oxygen Demand is a way of describing biochemical oxygen demand in terms of carbonaceous oxygen demand). Suppose that the environmental agency wishes to reduce the discharge from 200,000 pounds to 80,000 pounds per day, and that the daily total cost of treating q hundred pounds of waste is, say, \$.125 q^2 for X and \$.025 q^2 for Y[7]. A charge of 50 cents a pound would produce the desired elimination of 120,000 pounds of waste per day, since X would find it convenient to treat 20,000 pounds, and Y, 100,000 pounds (the total cost function for X is \$.125 q^2, his marginal cost is therefore \$.250 q. Thus, the marginal cost equals the charge when $q = 50/_{.25} = 200$; an analogous calculation gives the corresponding result for Y). In fact, as will be shown later, this pattern of waste reduction achieves the agency's quality goal of 80,000 pounds of FSUOD per day <u>at minimum cost</u>.

But the same result can be obtained also by means of the pollution rights scheme. If the agency issues 80,000 pollution rights, X will outbid Y for all of them, just as he finds it more profitable to pay the 50-cent tax on 80,000 pounds of his waste than to engage in costlier treatment.

More generally, if the number of pollution rights issued is less than the number of pounds of waste discharge (expressed in some equivalent measure), the rights will command a positive price and a continuous market will develop in response to the competition among buyers and sellers. This market will generate an equilibrium price at the point where the capitalized value of the marginal cost of treating an extra pound of waste just equals the price of the right. This means that the marginal costs of waste treatment will be the same for all dischargers in the same region, thus assuring that the agency's environmental quality objective will be satisfied at minimal total cost.

Although the two market approaches bring about the same result, in the simplified situation considered here the use of pollution rights presents several advantages over the charges system, especially when the pollution problem is viewed dynamically.[8]

It is now generally recognized that the main difficulty in internalizing the costs of environmental pollution is the ambiguous specification of exchangeable property rights in media like water and air; the creation of a market in pollution rights can be seen as an attempt to remove some of the ambiguity. Other schemes aimed at internalizing environmental externalities through utilization of the existing legal system, or a better definition and enforcement of property rights, have also been proposed. In fact, whenever the cost of reducing the damage inflicted by a pollution-creating activity (by decreasing the scale or changing the technology of the operation) is less than the benefits created by the abatement, there arises the possibility of a contractual solution. The pollutees can afford to pay the polluter enough to cover the abatement costs, and in such a way that the transaction is advantageous for all parties concerned.

It should be noticed that such a system of contractual payments (or "bribes," as they are often called) could in principle achieve the same result as a scheme of optimal effluent charges. The "in principle" character of this statement must be stressed, however. Unless the number of parties is quite small, the familiar free-rider phenomenon will appear. Since it is technically difficult to exclude anyone from the benefits of the pollution-abatement measures, the incentives to contribute to the necessary payments are correspondingly reduced. This will make the contractual solution infeasible without some assistance from the state to overcome the free-rider problem. Nor is this the only difficulty. How can one rule out the possibility that the contractual solution may degenerate into a form of blackmail, with one party creating pollution for the purpose of extorting payments from others? The answer is that polluters and pollutees are assumed to act within a legal framework that determines their respective bargaining position.[9] For instance, the existence of a right of a private individual to seek an injunction or to sue for the damage created by a pollution-generating activity will strengthen his bargaining position. But in a situation where one polluter inflicts relatively light damages on a large number of people, the bargaining advantage created by the right is more apparent than real, if nuisance actions are possible only on a case-by-case basis; for the costs of court action would probably exceed the expected benefits. Hence we can expect that a good deal of effort will be devoted to make class actions possible, or perhaps to introduce regulations forbidding certain types of activity (e.g., smoke ordinances).

Optimality of Pollution Charges

The existence of pollution poses an efficiency problem for the economy at large: failure to account for external costs leads to an oversupply of the pollution-producing good, and a corresponding undersupply of the benefits (e.g., good water and clear air) which are reduced by pollution. If the damage cost of pollution were internalized, resource use would become more efficient. As we have seen, efficiency can be restored, in theory, by imposition of a suitable charge.

In Figure 2, MD is the marginal damage cost curve and MA the marginal abatement cost curve. The efficient level of discharge is OE, corresponding to the point of intersection of the two curves. In order to induce a polluter to carry the abatement level to the point where discharge is reduced to OE, it is sufficient to impose a charge equal to OK per unit of discharge since now the marginal abatement cost over the range EA is less than the charge (OA represents the level of pollution in the absence of control measures).

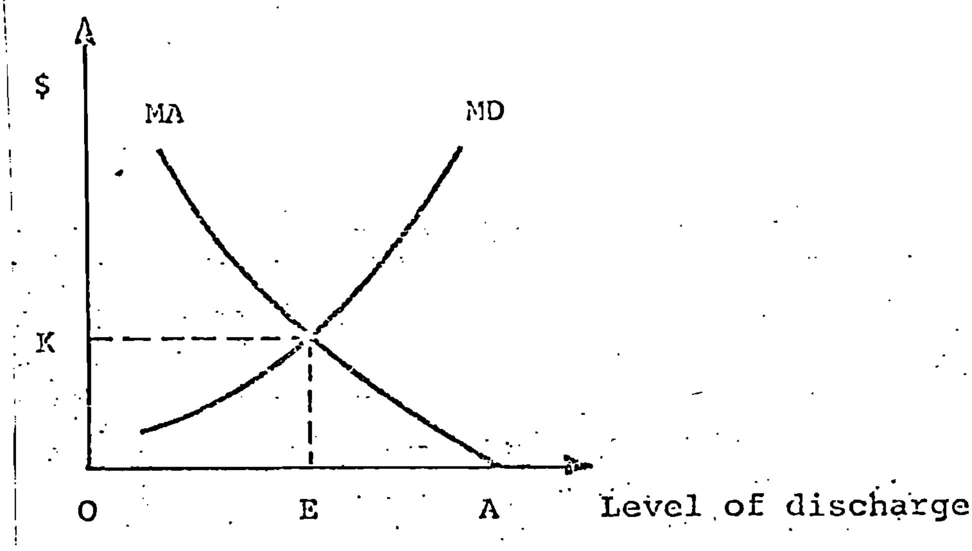

Figure 2

In the partial equilibrium context usually considered in problems of this kind, the determination of an optimal charge presents few theoretical difficulties. The practical problems, however, are staggering. A complication has already been mentioned in the preceding section: the optimal charge level is not equal to the initial marginal net damage, but rather to the damage that would result if the pollution-generating activity had been adjusted to its optimal level. Even more serious difficulties arise from the fact that neither abatement costs nor damage costs are known with sufficient precision. Abatement costs vary greatly among firms, municipalities, and households, and the sheer number of polluters makes reliable estimates all but impossible to obtain. The situation is, if anything, even more hopeless in the case of damage functions, where we must reckon with the practical impossibility of quantifying some of the basic values affected by pollution. However, ecological damage varies not only with the type and level of activity, but also with the location of the polluting unit.

Since the direct route to an optimal system of pollution charges is beset by too many obstacles, an alternative second-best approach seems more promising. It consists in assuming a given set of ambient standards (i.e., the standards are taken to be determined exogenously through the political process), while the price system is put to use in the realization of the environmental constraints which "society" imposes on its activity. Specifically, the charges (prices) are to be selected so as to achieve given acceptability standards at minimum cost rather than attempting to base them on the unknown value of marginal net damages.

In the case of water pollution, for example, this amounts to taxing all installations discharging waste into a river at a rate t(f) cents per gallon, where the tax rate t would depend on the FSUOD value of the effluent, according to some fixed schedule. Mistakes in setting the charges would now be easily detected from the deterioration of the quality standards (if the charges are set too low), and corrective action taken (even this is not as easy as it sounds, since expectations have been based on the previous set of charges; but there is no doubt that the difficulties here are orders of magnitude smaller than those facing a pure system of pollution charges). This charges-and-standards approach enjoys a significant optimality property: it minimizes the social cost of achieving the desired reduction in the total emission of pollutants. An intuitive proof of this proposition is easily given. Suppose it is decided that the SO_2 content of the atmosphere in a certain region should be reduced by 50%. The environmental agency places a unit charge on smoke emissions, raising the level of the tax until SO_2 emissions are in fact reduced by 50%. In response to these charges, polluting firms (which are here assumed to be cost-minimizers, though not necessarily profit maximizers) will cut back their smoke emissions until the marginal cost of further reductions is equal to the charge. Since the charge is the same for all installations in the area, marginal costs of reducing smoke output will be equalized across all activities. But this implies that it is impossible to reduce the aggregate (social) cost of achieving the specified quality standard by rearranging smoke-reduction quotas: any alteration in this pattern of smoke emissions would involve an increase in smoke output by one firm, the value of which to the firm would be less than the cost of the corresponding reduction in smoke emissions by some other firm. For a formal proof, see Baumol and Oates.[10]

Efficiency vs. Equity

To make the ever-present conflict between efficiency and equity considerations as clear as possible, it seems advisable to start with a simple numerical example. Let us go back to the case already considered of two polluters, X and Y, each discharging into a river raw sewage equivalent to, say, 100,000 pounds of FSUOD a day. The environmental agency wishes to reduce the discharge from 200,000 to 80,000 pounds/day. To achieve this goal, a number of methods can be used, ranging from pollution charges or the issuance of pollution rights, to a variety of cut back patterns imposed by direct orders. Thus, X could be required to eliminate all of its waste, while Y would eliminate only 20,000 pounds, or conversely. In practice, elementary considerations of equity seem to dictate that if a regulatory approach is used, each polluter should treat waste to the same extent; in the example, X and Y would be required to eliminate 60,000 pounds of waste each.

But polluters differ with respect to their treatment costs. Let us say that the daily total cost of treating q hundred pounds of waste is, as before, $\$.125q^2$ for X, but only $\$.025\,q^2$ for Y, so that marginal cost of treatment rises five times faster for X than for Y. Taking differential costs into consideration, the simple notion of equity used in arriving at the equal allocation of pollution abatement effort to the two polluters (50%) reveals its limitations. For the costs of "equal" effort are $(.125)\ (600)^2 = \$45{,}000$ for X and $(.025)\ (600)^2$ \$9,000 for Y.

Requiring equal financial effort would perhaps be a more equitable way of proceeding. Before discussing this possibility, let us derive the least-cost (efficient) solution to the agency's pollution control problem. The total cost function is given by

$$C = 1/2 \left\{ .25(aq)^2 + .05 \left[(1 - a)\, q \right]^2 \right\}$$

where $0 \leq a \leq 1$ is the proportion treated by X, and q is the total load treated in hundreds of pounds.

Since $\frac{dC}{da} = .25\, a\, q^2 - .05\,(1 - a)\, q^2 = .30\, a\, q^2 - .05\, q^2$, the minimizing value is $a = \frac{.05}{.30} = \frac{1}{6}$.

The environmental standard prescribes the value $q = 1200$. Therefore, in a least cost solution X should eliminate 20,000 pounds of waste at a cost of \$ .125 $(200)^2$ = \$5000, while Y is required to eliminate all his waste at a cost of \$ .025 $(1000)^2$ = \$25,000. Any other allocation would increase total costs, since it involves requiring X to treat a pound of waste when Y could do so at less cost.

But equity demands that two individuals who have jointly caused a harm in equal measure should make an equal effort to eliminate the harm. Of course, as already mentioned, the meaning of "equal effort" is ambiguous: it may be interpreted as meaning that each polluter should treat the same percentage of his waste or, alternatively, that each should make the same financial sacrifice. Neither interpretation leads to the least-cost solution; though equity and efficiency could, in principle, be reconciled using a two-stage procedure.[11] Consider first the principle of equal financial sacrifice. Step one: the agency determines the least-cost solution which imposes treatment costs of \$25,000 to Y and \$5000 to X. Step two: X is induced to pay Y \$10,000.

A similar procedure could be used to reconcile efficiency with the principle of equal percentage of treatment. First, the agency calculates the cost of a uniform percentage plan meeting its quality standards. Second, the agency issues orders to X and Y according to the least-cost allocation, but would also require monetary transfers to achieve a proprotionate division of costs equivalent to that which uniform treatment would produce. In our example, uniform treatment requires 60% removal for both X and Y, at a cost of \$1/2 (.25) $(600)^2$ = \$45,000 and 1/2 (.05) $(600)^2$ = \$9,000, respectively. Thus Y pays 1/6 of the total cost (\$54,000) of the uniform treatment scheme, whereas in the least-cost solution his cost is \$25,000 (out of a total \$30,000). Since one-sixth of \$30,000 is \$5,000, X must pay Y \$20,000.

Unfortunately, institutional constraints seem to rule out such possibilities of reconciling efficiency and equity (this difficulty is well known to welfare economists). Basic regulatory structures control the extent to which cost minimization may be achieved and the way in which the clash between fairness and efficiency is perceived and resolved. For instance, an agency adopting the market solution confronts the efficiency and equity questions in somewhat different terms than an agency using a regulatory approach. In the former case, the crucial decision concerns the determination of the "right" price of pollution; whatever measures are then considered in order to reconcile the resulting allocation with equity criteria, they must not distort market incentives. In the latter case, uniform percentage reduction can be simply mandated, even without knowledge of abatement costs.[12]

Limits of Environmental Deregulation

The traditional approach to environmental policy is characterized by "an almost exclusive reliance on detailed central regulation and on court enforcement as the principal techniques for limiting the discharge of pollution."[13] Bureaucratic rigidity, economic inefficiency, lack of incentives to technical innovation, and a proclivity to political manipulation and bargaining are frequently mentioned causes of the unsatisfactory performance on the traditional system. Deregulation -- greater use of markets and the price system -- has been advocated as the appropriate remedy. But proof of "non-market failure" is not sufficient evidence that one should turn to the market. As Charles Schultze reminds us, "In all cases, the comparison should be between an imperfect market and an imperfect regulatory scheme, not some ideal abstraction."[14]

A realistic comparison of effluent charges with standards reveals that the advantages of the former system are less impressive than abstract economic arguments seem to imply. First, as already indicated, computational problems and difficulties in measuring many important benefits and costs preclude the possibility of setting charges at the "correct" level, where the marginal costs of discharge reduction at each source just equal the marginal social damages caused by that source. Second, as Clifford Russell has argued,[15] a system of uniform charges is almost certainly incompatible with discharge-standards target. Thus, an environmental agency would have to perform a separate calculation for each individual source of pollution (there are about 60,000 such sources in the United States). In fact, each price would have to be calculated several times by the very nature of the trial and error method used in arriving at the proper charge. Third, there is little to choose, in principle, between charges and standards in terms of static efficiency since an efficient set of effluent charges may be viewed as the dual solution of the problem of determining the discharge standards minimizing the cost of achieving given ambient quality standards. This applies, in particular, to the information needed to solve either the primal or the dual problem. Also monitoring and implementation problems seem to be of approximately equal levels of difficulty in the two cases.[16]

Finally, there is no reason to believe that a system of effluent charges would be less open to political pressures and bargains than a direct regulatory approach, as so often suggested by advocates of charges. All the available empirical evidence indicates that the setting of charges is as much a political process as the setting of standards.[17] Effluent charges do enjoy one clear advantage over direct regulation: by lifting air and water out of the category of free goods the system creates important incentives for polluters to change their technologies in ways that would economize on the use of the environment. This is, of course, an important property; but it is equally important to realize that there are many situations where the logic of marginal comparisons of benefits and costs is not even applicable.

Consider the case in which no negative external effects seem to be present, simply because people are unaware of the dangers of the situation. For example, a number of heavy metals like cadmium or mercury are highly toxic but their effects are detected only after they have accumulated, in kidneys or other human organs, above a certain threshold. Here economic analysis would suggest a status quo policy since no externalities are present, whereas biological criteria recommend positive action. Moreover, in the case of non-degradable pollutants like cadmium, and many types of radiation, the damage arises from an essentially non-reducible <u>stock</u>, so that only incremental damage results from the <u>flow</u> of pollution. Now, one of the crucial assumptions in a cost-benefit or market approach to pollution control is that the pollution level may be varied in any direction. However, in the case of a non-degradable pollutant, the marginal damage function can only move in an upward direction, since the stock of pollutant already present in the environment cannot be reduced within time periods that are meaningful for policymaking.

Especially in a dynamic context, environmental standards may be definitely preferable to regulation through prices. Consider a situation in which the volume of waste exceeds the assimilative capacity of the environment, and suppose that a Pareto-optimum level of pollution, denoted by l_0, has been determined on the basis of a cost-benefit calculation. Unless $l_0 = 0$, a reduction in the assimilative capacity of the environment will result as the counterpart degrader population (on which the assimilative capacity of the environment depends) is reduced in size. Thus, as the assimilative capacity level drops, a new optimum pollution level has to be determined, corresponding to a new (and steeper) marginal treatment cost curve. As long as the volume of waste exceeds the assimilative capacity of the environment, the situation will remain ecologically unstable, and if continuous Pareto adjustments are made, zero output will be the limiting solution.[18] However, this result can be avoided if a standard is set at the very beginning, fixing it at the output level where the waste generated equals the initial assimilative capacity of the environment. Thus, "where the context takes on dynamic externality features or where biologically harmful pollutants cumulate, cost-benefit should give way to standard setting based on a cautious attitude to epidemiological and other physical information."[19]

The Uncertain Logic of Standard Setting

But is a "cautious attitude to epidemiological and other information" sufficient to provide a rational basis for standard setting? Standards cannot be established on purely scientific and technical grounds any more than values can be deduced from facts. In the past, standard setters have sought legitimacy for their decisions by wrapping them in a cloak of scientific respectability, but the courts and informed public opinion are becoming too sophisticated to be so easily misled. For example, a section of the 1963 Clean Air Act instructed the Secretary of HEW to "compile and publish criteria reflecting accurately the latest scientific knowledge useful in indicating the kind and extent of such effects which may be expected from the presence of such air pollution agents (or combination of agents) in the air in varying quantities." In the next section, however, political judgment rather than scientific knowledge is the guiding principle: "The Secretary may recommend to such air pollution control agencies. . . such criteria of air quality as in his judgment may be necessary to protect the public health and welfare." This ambiguity in the conception of what environmental quality criteria should be was to be the source of considerable controversy.[20] In particular, the first standard for SO_x, establishing a recommended level of annual average concentration of 0.015 ppm, was greeted by a storm of controversy. The scientific basis looked impressive at first sight -- the main part of the document consisted of summaries of almost 350 studies on the effects of SO_x on man, animals, plants, and materials. But critics were quick to point out that not all the evidence was equally reliable; that the summaries of the studies drew conclusions which the studies themselves did not; that the recommended level was based on only two or three studies whose conclusions were, moreover, open to doubt; and, finally, that the derivation of the standard from the goal of protecting the most susceptible segments of the population had not been made explicit. In part as a consequence of such criticism, the 1970 Amendments to the Clean Air Act introduced the notion of threshold value -- a level of ambient concentration below which it is assumed that no damage occurs to health. Congress directed EPA to use "scientific evidence" to determine threshold values for pollutants assumed to have them (SO_2, CO, NO_x, particulates and oxidants), and to take those values minus "an adequate margin of safety" as primary standards. But despite the appeal to scientific evidence, the notion of threshold value has been aptly described as a "politically convenient fiction which permits the law to appear to require pollution damage to be reduced to zero."[21]

Regulatory decisionmaking in other health-related areas also relies heavily on threshold values. For example, food containing a potential carcinogen, but which does not enter the food supply as an additive governed by the Delaney clause,[22] may be sold only if levels of the substance are below a certain value, termed a tolerance. How is the acceptable daily intake (ADI) determined in practice? The standard toxicologic procedure involves the notion of a "no observed effect level" (NOEL), which EPA has defined as "the level (quantity) of a substance administered to a group of experimental animals at which those effects observed or measured at high levels are absent and at which no significant differences between the group of animals exposed to the quantity and an unexposed group of control animals maintained under identical conditions is produced."[23]

Other methods of risk assessment assume the absence of a threshold, extrapolating downward from the observed level to a risk level considered virtually safe. Many mathematical dose-response functions may be used in the extrapolation. Unfortunately, while the choice of function has a major effect on the "virtually safe dose" (more than 100,000-fold, according to the Food and Drug Administration Committee on Safety Evaluation[24]), there is no firm scientific basis at present for choosing among them. Even the convenient assumption of low-dose linearity lacks convincing scientific support, as shown by the intensity of the controversy following the attack by Gofman and Tamplin on the radiation standards used in the 1960's (the subsequent tightening of those standards has been made possible by technological advances rather than by any significant improvements in laboratory results, or in statistical and mathematical techniques). After carefully reviewing different methods of carcinogenic risk assessment, Jerome Cornfield comes to the conclusion that:

> Although many observed dose-response curves are consistent with the existence of thresholds. . . no finite set of dose-response observations could establish this. All present safety evaluation procedures, whether involving the use of NOEL's, or of some favored non-threshold dose-response function with a "virtually safe" level, must be regarded as mathematical formalisms whose correspondence with the realities of low-dose effects is, and may long remain, largely conjectural. But regulatory decisions must be made, and formalisms with more theoretical or experimental support, or both, should be preferred to those with less.[25]

Who should evaluate the evidence? The regulatory decisionmaker assisted by his advisors? Independent experts? Scientific "judges?" And which procedures and rules of evidence ought to be used? Such questions are implicit in the widespread demands for greater public access to the decisionmaking processes of regulatory agencies and for improvements in the quality of agency deliberations. Before discussing some possible answers, it may be useful to examine the underlying epistemological issues.

Trans-scientific Issues in Standard Setting

Disagreement among experts is a pervasive characteristic of many policies with important technical or scientific components. Expert A disagrees with the arguments presented by expert B, but finds it difficult or impossible to disprove specific points. Both experts soon come to recognize that the complex technical issues about which they must express an opinion require perceptions that cannot be explicitly articulated, and patterns of reasoning that are alien to their disciplinary traditions. For example, most scientists who have worked on setting radiation standards have severely criticized the arguments used by Gofman and Tamplin in support of a ten-fold reduction in federal standards without, however, being able to demonstrate clear-cut technical errors in their analyses. Rather, the criticism has been directed at the plausibility of the assumptions made by the two researchers, at the alleged arbitrariness of their selection of data, and at their "lack of professionalism."[26]

Problems relating to the determination of the health effects of low-level radiation are examples of questions which Alvin Weinberg has termed "trans-scientific"; questions of fact that can be stated in the language of science but are, either in principle or in practice, unanswerable by science. As Weinberg points out "In so far as public policy involves trans-scientific rather than scientific issues, the role of the scientist in the promulgation of such policy must be different from his role when the issues can be unambiguously answered by science."[27]

When different scientists put different interpretations on, or draw different conclusions from the same body of data; when they disagree about the relevance or strength of the evidence and the plausibility of hypotheses; when their preference for a given model (say, the linear model of dose-effect for radiation exposure) is based not so much on technical grounds as on the fact that it is the most conservative model for purposes of public safety; when arguments are evaluated not on their intrinsic merits, but by appeal to the majority of professional opinion, and the credibility of the opponent becomes at least as important as his competence -- we feel that the generally accepted paradigm of scientific inquiry is no longer applicable, and that we are dealing with a new game whose rules are not yet clearly understood, let alone explicitly formulated. The amazement of the policymaker who discovers how elusive the notion of "scientific evidence" can be, and how discretionary the choice among competing models, matches the alarm of the traditional scientist who fears that the principles of scientific method may be contaminated by the atmosphere of the adversary process.[28]

The uneasiness of the scientist is understandable; for the notion that truth, or at least agreement, may emerge from the clash of conflicting opinions marks a significant departure from beliefs and attitudes that have dominated scientific thinking for more than three hundred years. In a famous passage of his Regulae ad Directionem Ingenii, Descartes wrote: "Every time two men make a contrary judgment about the same matter, it is certain that one of them is mistaken. What is more, neither of them possesses the truth, for if one of them had a clear and precise view of the truth, he would be able to expound it to his opponent so as to force the latter's conviction." Bacon demanded that all preconceived notions, opinions, even words "be abjured and renounced with firm and solemn resolution," and condemned "disputation" -- the art of dialectic argument created by the Greeks and further developed by the scholastic philosophers. Similarly Galileo, in the words of Ernst Mach, "with a superb indifference to the dialectic arts and sophistic subtleties of the Schoolmen of his time, turned the attention of his brilliant mind to nature."[29] Modern scholarship has shown that Galileo and other great scientists of the past were well aware of the value of persuasive arguments and of a skillful use of evidence in winning scientific battles.[30] But in the context of the present discussions, this is not as important as the fact that in the prevailing opinion, scientific disputes are resolved by interrogating nature rather than by appealing to human judges.

Actually, the belief in the power of science to provide unambiguous answers to precise questions is more characteristic of people without a scientific background than of working scientists. Many legislators and administrators expect unqualified yes-or-no answers from their scientific advisors, and it is often asserted that open disagreement among experts can only confuse policymakers. Yet those same policymakers can tolerate high levels of conflict and ambiguity in dealing with political and social issues. Is it because policymakers are often unwilling to make the effort necessary to master the details of complex technical arguments? Perhaps, but significant attitude changes can be expected to take place as public criticism continues, and as reviewing courts subject to more searching scrutiny the empirical and analytical basis for an agency's discretionary policy choices, and measure its justifications against the recorded evidence.

Adversary Procedures

Since many of the technical issues arising in environmental standard setting are trans-scientific, the resolution of conflict concerning scientific evidence and analyses is a problem of crucial importance to the regulatory bureaucracy. Efficient procedures of conflict resolution must reflect the cognitive characteristics of trans-scientific issues. Since these issues are debated by opposing arguments and argumentation is essentially a dialectic skill, an adversary procedure appears to be the natural format for trans-scientific debates. Adversary procedures are "those formal, legal or quasi-legal proceedings at which proponents, both scientists and non-scientists, of opposing views are heard before a body or an individual who is empowered to render a decision after having heard the conflicting contentions."[31]

The procedure provides incentives for the adversaries to present the strongest arguments in support of their respective positions, while disciplining them by cross-examination and by forcing them to answer the specific questions posed by the judges. The assignment of precise roles facilitates the orderly flow of information and the systematic, step-by-step construction of the "case." The excesses of demagogy that are common in less structured debates are prevented by the rules of procedural etiquette. If these represent the more obvious advantages of the adversary procedure, what are seen as its shortcomings? The following list includes, I believe, the most common objections:

1. It is not an appropriate method for discovering truth
2. It paralyzes decisionmaking
3. Instead of encouraging convergence, it intensifies the polarization of viewpoints
4. It discriminates against those who do not command adequate expertise
5. It can be manipulated in favor of the financially or politically powerful
6. Legislators and administrators cannot be expected to immerse themselves in the technical details of a case as a judge does.

The rationale for the last statement is not clear. In the case of tax laws and some of the more recent environmental legislation, for example, legislators have shown considerable ability in handling highly technical subjects. In fact, carefully structured adversary proceedings represent a natural sequel to the traditional system of congressional hearings. On the other hand, as I have already indicated, regulatory and even executive agencies have come under pressure from the courts to set out their decisions in "proper form," by introducing formal hearings with sworn testimony, rights of cross examination, judicial review of the evidence, and so on. If it is true that "The adversary procedure is likely to be used increasingly in modern, liberal societies in their attempts to weigh the benefits and risks of modern technology,"[32] we can expect the present trend toward more formalized procedures in regulatory decisionmaking to gain momentum in the future.

Of course, the opportunity costs of the added procedural constraints can be high in terms of delay and administrative complexity. In the United States it now takes ten or twelve years to bring a nuclear power plant from the planning stage to commercial operations -- nearly twice what it took a decade ago. Yet, when the technical and value basis of a regulatory decision is controversial, such costs should be reduced by improvements in procedural design rather than by an expansion of the agency's discretionary power. In fact, this philosophy underlies recent proposals for streamlining the licensing process of nuclear power plants by limiting the number of times the same objection can be raised, and granting utilities prior approval for power plant sites and standardized reactor designs. True, the amendments to the 1972 Clean Water Act currently being considered by the U.S. Congress seems to move in the opposite direction, since they eliminate formal hearing requirements and give additional discretion to the EPA administrator. Whether the short-run efficiency induced by these measures will be sufficient to compensate the increased likelihood of subsequent litigation remains to be seen. In more general terms, the widespread view that regulatory decisionmaking is already too burdened by procedural requirements is not supported by the evidence. Thus, a recent survey of twenty-two major federal regulatory agencies found that in only nine of them is expert cross-examination of experts permitted (usually, under various restrictive conditions), and in only one agency -- the International Trade Commission -- cross-examination of experts occurs fairly often. The Environmental Protection Agency does not permit cross-examination, while the Nuclear Regulatory Commission permits it, but uses it only seldom.[23]

Procedural rules can be manipulated in favor of special interests, but so can any other institutional arrangements. If anything, the openness of adversary proceedings should make manipulation more expensive and at the same time more risky. It is also true that differentially distributed resources (including money and technical expertise) are valued differently under alternative institutional arrangements. The problem of assuring that all parties are effectively able to make use of the procedural possibilities open to them is a real one, but this is a pervasive problem, not one peculiar to a particular procedure. Naturally, a regulatory agency should try to redress any inequality in the positions of the contending parties, and several methods for achieving this have been discussed in the literature. The two remaining objections to the use of the adversary procedure -- that it is not a suitable method for determining truth, and that it intensifies the polarization of viewpoints -- will be evaluated in the following pages.

The Proposed Science Court

It is important to make a clear distinction between adversary proceedings and the science court recently proposed by a number of scientists. The science court is only one among several possible institutional embodiments of the adversary procedure. Many scholars who favor an expanded use of the latter method have reservations about the usefulness of a science court.

As proposed by the Task Force of the Presidential Advisory Group on Anticipated Advances in Science and Technology in 1976, the science court is to be concerned solely with questions of scientific fact, not of social value. The goal is

> To create a situation in which the adversaries direct their best arguments at each other and at a panel of sophisticated scientific judges rather than at the general public. The disputants themselves are in the best position to display the strengths of their own views and to probe the weak points of opposing positions. In turn, scientifically sophisticated outsiders are best able to juxtapose the opposing arguments, determine whether there are genuine or only apparent disagreements, and suggest further studies which may resolve the differences.

In this way, it should be possible "to describe the current state of technical knowledge and to obtain statements founded on that knowledge, which will provide defensible, credible, technical bases for urgent policy decisions."[34] There are three stages in the science court proceedings. First, the significant scientific and technological questions associated with a controversial policy issues must be identified. By definition, an issue is a decision pending before a governmental agency. However, the science court would not consider the entire issue, but only what it considers its most important (and controversial) scientific and technical aspects.

The second stage is represented by an adversary hearing, open to the public, governed by a disinterested referee, in which scientist-advocates of the opposing scientific positions argue their cases before a panel of scientist-judges. In addition to presenting their cases, the advocates have the opportunity to cross-examine opposing advocates and to criticize their arguments. The judges themselves are established experts in areas <u>adjacent</u> to the dispute.

In the third stage, after the evidence has been presented, questioned, and defended, the panel of judges issues a report in which the points of agreement among the advocates are noted, and judgments on disputed statements of fact are given. The report may also suggest specific research projects to clarify points that remain unsettled.

These, in brief, are the goals and modus operandi of the proposed science court. A three-day colloquium on the proposal held in the Fall of 1976 and attended by 250 lawyers, administrators, industrialists, and scientists and other academics, raised considerable interest. In the opinion of its organizers, the meeting was to be the first experimental test of the science court idea although, in the words of a participant, "proponents appeared to be well committed to what seemed more like a demonstration project than an experiment."[35]

As the guidelines make clear, the court (or "quasi-court") will only examine, and decide upon, questions of scientific fact, where by "scientific fact" is meant "a result, or more frequently the anticipated result, of an experiment or an observation of nature."[36] These statements raise a number of problems. First, the court must select, from all the scientific and technical questions relevant to the policy issue under consideration, those it considers most significant. But this is obviously a value judgment, not a factual statement. Even the separation of the technical from the political or ethical aspects of the problem may imply a more or less conscious value judgment.[37] A similar question -- the separation of "efficiency" from "equity" considerations -- has been debated for a long time by economists. Today a majority of professional opinion feels that in a policy context such a separation is either impossible, or strongly biased in favor of the status quo.

The same arguments apply, mutatis mutandis, to the fact-value dichotomy in the present context. (Incidentally, the epistemological premises underlying the science court idea also imply a sharp separation of facts from theories. But according to many contemporary philosophers of science, facts are "theory-laden," since in all the experimental techniques of the scientist, theories are involved.)

Second, the narrow notion of scientific fact adopted by the proponents of the science court would exclude from its proceedings all questions of a trans-scientific nature. For example, in setting radiation standards it is quite important to decide whether or not the genetic response to low-level radiation is linear. However, to determine at the 95 per cent confidence level by direct experiments whether 150 millirems will increase the mutation rate by ½ per cent would require about 8×10^9 mice, so that in practice, the question is unanswerable experimentally.[38] To take another example, consider the following proposition: "An acceptable version of the statement ["if X occurs, then Y may occur"] must specify a finite probability which could be refuted by a possible experiment."[39] Suppose one expert claims his calculations show that the total probability of a catastrophic reactor accident is of the order of 10^{-7}/reactor/year. With such a small probability there is no practical possibility of determining this failure rate experimentally. One can only use various mathematical, statistical, and analogical arguments to evaluate the plausibility of the calculation. Now, if trans-scientific questions like the ones just mentioned are ruled out of court, why are adversary proceedings among scientific experts at all necessary? As long as the facts can be established by the usual procedures of science, there is little need for scientist-advocates, and even less for scientist-judges.

Concerning evidence, the report of the task force states that "the applied rules of evidence will be the scientific rules of evidence and not the legal rules of evidence." But while there is an elaborate law of judicial evidence, scientists have not found it necessary to codify the rules of scientific evidence. One reason for this apparent paradox has to do with the quality of the evidence itself: complex and often ambiguous in the law, relatively simple and usually of high quality in the sciences. Moreover, the law of evidence and indeed the entire legal system, reflect the obligation of the judge to give judgment in any circumstances, while no such obligation is imposed on the scientist. Now, complexity and ambiguity of the available evidence, and the need to reach a conclusion in all cases are also characteristic features of regulatory decisionmaking. This is not meant to suggest that the law of evidence is immediately applicable in debating trans-scientific questions; rather, that unarticulated rules of scientific evidence are not sufficient, and that a more formal approach is needed. But then the idea of scientist-judges loses much of its plausibility. As physicist Barry M. Casper writes:[40]

> The notion that a scientific background is required to weigh the claims of experts is difficult to reconcile with the evidence of recent debates such as those over the ABM and the SST. It is not necessary to master the detailed workings of these systems in order to judge even the technical points at issue if one has the opportunity to hear articulate advocates present their cases and respond to opposing arguments. A careful reading of the public records of the ABM and SST debates indicates that there is rarely significant disagreement over "scientific facts." When apparent disagreements over these facts occur, they generally can be traced to differences in assumptions.

Paul Doty has pointed out that in order to make the contributions of technical experts more relevant to the policy process, a reasonable balance of attention should be given to both the arguments that depend on detail and the wider issues that depend on experience, judgment, and perception of the political context.[41] Only those responsible for the final decision -- the regulators themselves, rather than scientist-judges -- can provide this necessary balance.

Conclusions

The preceding discussion makes clear that in the environmental field, as in other areas of public policy, there are no unambiguously superior policy tools. In devising a comprehensive approach to pollution control, it would be foolish to forego the advantages associated with a system of effluent charges, particularly in connection with the question of incentives for technological innovation. But it would be an even more serious mistake to believe that the prevailing regulatory approach can be completely replaced by charges and other price-like mechanisms. Environmental deregulation -- if by this is meant a system of market incentives which is rich enough to cover every kind of pollution problem -- is not a feasible alternative. At any rate, the conflicting demands of efficiency and equity can only be reconciled by political means. Hence, whatever the technical properties of the tools that are used, actual policy results will be largely conditioned by political pressures and bargains.

Direct regulation is, and will remain, an essential component of environmental policies. This statement can be supported by three types of considerations: institutional, administrative, and informational. Institutionally, the process of standard setting is well adapted to political and bureaucratic realities in a way in which pollution charges and rights, for example, are not. Administratively, direct regulation is unavoidable when quick action must be taken in response to sudden emergencies or to environmental situations that may result in irreparable damages. From the point of view of the information requirements of environmental policymaking, standards -- being essentially empirical guidelines summarizing the available scientific, technical, and economic evidence -- can be used without the precise knowledge on costs, damages, and benefits that is required by more sophisticated methods.

Certainly the rationality of the standard-setting process leaves much to be desired. This has led many critics to advocate the abandonment of standards as the principal method of pollution control. A more realistic approach consists in attempting to improve current practices without rejecting their basic logic -- at least until the viability of alternative methods can be established beyond reasonable doubt. Major improvements seem possible in at least two directions, through better procedural design. First, more attention should be given to different methods of achieving alternative distributions of bargaining power among the different participants in the policy process. One method, exemplified by the U.S. National Environmental Policy Act, is to empower any citizen or private organization to sue any private or public body for its environmentally harmful actions. Another possibility is class action in courts. When individuals or organizations can sue on behalf of large numbers of similarly affected citizens, the benefit-cost terms of using the courts are significantly altered in favor of the damaged parties. Second, the utilization of empirical data and scientific knowledge by regulatory agencies can at best be characterized as haphazard. This has led the courts in the United States, but apparently also in Germany and Japan, to impose on regulatory agencies strict procedural requirements for the evaluation of evidence, the representation of conflicting opinions, and the examination of a richer set of alternatives than were originally considered by the regulators. In the United States this has led to a "paper hearing" procedure that combines many of the advantages of a trial-type adversary process (excepting oral testimony and cross-examination) while avoiding undue delay and cost.

There is considerable evidence to show that the requirement that a regulatory agency explain in detail the factual and methodological bases of its conclusions can significantly improve the quality of environmental decisionmaking.

What is even more significant, an open record that encompasses the evidence and conclusions of the agency and its opponents, and includes responses by the agency to criticism of its decisions, can considerably increase the legitimacy of those decisions.

When the truth, correctness, or fairness of a decision can be determined unambiguously, the manner in which the decision is reached is largely immaterial; only results count. But when the factual and value premises are uncertain or controversial, the formal characteristics of the decision process -- its procedures -- assume a new significance. Legal or quasi-legal procedure is not so much a criterion of truth as a device to ensure that acceptable decisions are reached, whether or not solutions can be calculated in a logically impeccable manner. A system that must guarantee the decidability of all emerging issues cannot at the same time guarantee the correctness of each individual action. For this reason, the crucial problems facing environmental policymakers today have to do more with good procedural design than with the search for strategies satisfying abstract criteria of optimality.

Notes

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11. B.A. Ackerman et al., op. cit., pp. 231-233. This discussion of the equity issue closely follows arguments presented by Ackerman and co-authors.
12. In practice, some information on abatement costs will be needed in order to set reasonable fines for non-compliance.
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15. Clifford S. Russell, "The Characteristics of Effluent Charge Systems," Policy Analysis, forthcoming.

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21. Allen V. Kneese and Charles L. Schultze, op. cit., p. 51

22. According to the Delaney clause (U.S. Code 21, 348 (c) (3)), ". . . no additive shall be deemed safe if it is found to induce cancer when ingested by man or animal, or if it is found, after tests which are appropriate for the evaluation of the safety of food additives, to induce cancer in man or animal. . ."

23. See Jerome Cornfield, "Carcinogenic Risk Assessment," Science, 18 November 1977, vol. 198, No. 4318, pp. 693-699.

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A POLICY FRAMEWORK FOR THE STANDARD-SETTING PROCESS*

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A POLICY FRAMEWORK FOR THE STANDARD-SETTING PROCESS

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CONTENTS

A POLICY FRAMEWORK FOR THE STANDARD-SETTING PROCESS

INTRODUCTION

The standard-setting process has a socio-economic and political basis. This statement has been made previously by Majone and Holden. Majone has noted the following points in line with this thesis [1]:

- a standard cannot be set on a purely scientific basis;
- a standard provides only an appearance of precision and hence of "scientific" character;
- a standard always represents an implicit evaluation of human well-being;
- a standard is only one of other alternative means of regulation;
- the institutional framework often determines the decision on a standard;
- self-interest of regulatees moves them to attempt to modify the terms of the regulator, including any standards set.

In addition, Holden has noted that regulatory processes, including standard-setting, are based on a bargaining process between regulators and regulatees [2]. As an example of the political nature of standard-setting Schon notes that an attempt to set standards in the lumber industry for the "2x4" developed more political response than any other issue in the recent history of the U.S. Department of Commerce [3]. Lumber producers both large and small, building interests, federal agencies, state governments as well as U.S. Congressmen and Senators were all engaged in attempting to influence the thickness standard of the "2x4". Schon posits that a system of "dynamic conservatism" builds up around a certain technology that exists to protect that technology whenever it is threatened with, say, standards that might initiate any loss of market, influence or position [4].

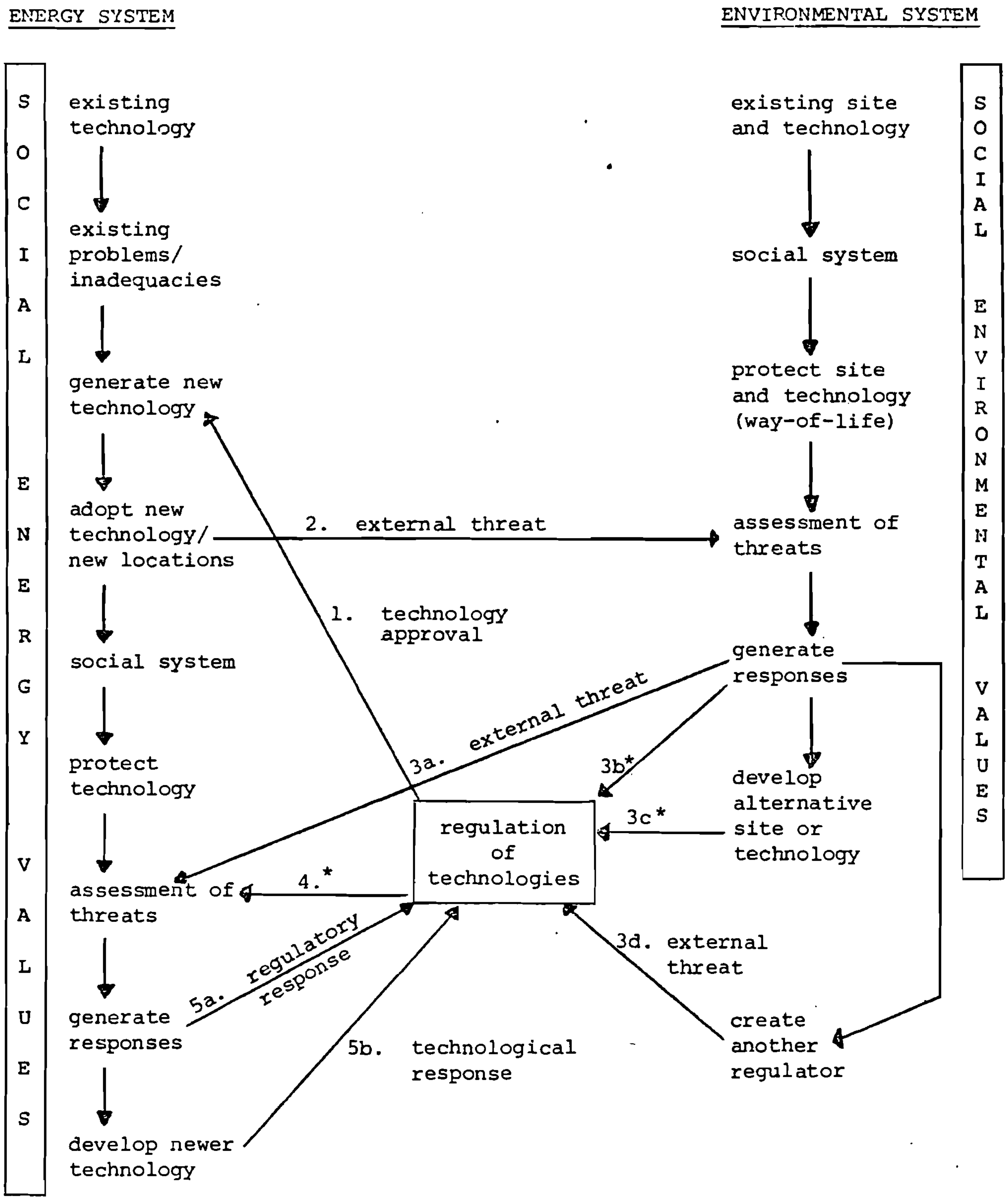
A key attribute of energy production as an issue which arouses and focuses attention is its location at a particular site. Both the need for energy development and its attendant benefits and costs are perceived in local terms, even though such development has large overall national benefits and costs as well. What heightens the energy issue even more at the local level is the use of nuclear power which is not restricted by location but is shiftable among a variety of locations, while petroleum or coal must be developed *in situ* where it is found. Wherever a nuclear power station is placed both benefits and risks are heaped on a particular site regardless of the overall benefit-cost ratio to the nation.

Both the site and the energy technology appeal to the "protective instincts" of the particular social groups or actors that array themselves around each component. Those local actors who stand to gain from the development of energy at a site ally themselves with those interested in embedding and extending that energy system in the overall economy. Those local actors who do not perceive direct economic benefits or share energy values stemming from that energy system (say, nuclear) will see themselves as bearing the total risk of such a development and oppose it. Those oriented to the site tend to be those motivated by environmental and traditional technology values well embedded in the existing social system. The premise for conflict tends to be that if those in the locality do not pursue their own interests then their interests will be ignored by those at the national level who are more oriented to the technology than to the site. What precipitates an incident or conflict is a specific proposal to place a nationally approved technology on a particular site. This proposal places site-oriented and technology-oriented values into conflict where each set of actors attempts to protect the values inherent to the social systems surrounding the technology and the site. This emphasis on a specific project at a specific site tends to shortcut broader discussions of goals and alternative means.

TECHNOLOGIES, ENVIRONMENTS AND THREATS

In an attempt to display the basic processes involved in the protection of a technology and a site Figure 1 was constructed. It is a process that does not try to model the protection process per se but rather attempts to simply portray an overview of key aspects in local actors responding to a threat to some technology through an effort to regulate it.

At any one time society has both energy and environmental values and needs that must be met through whatever technologies and sites that exist. The existing energy production system embodies all of the technology and social value systems necessary to perpetuate that production source over time in meeting the energy demands of the economy it serves. The existing environmental system also embodies all of the technology and social value systems necessary to maintain the local employment and living base over time at the existing locale. In the case of environmental values such awareness can grow around a traditional technology supporting some existing economic activity on a site sustaining a way-of-life such as single-family fishing dories or trawlers. The existing technology becomes traditional through the social system that builds up around it at that site. Often an entire community will be based on the one existing technology. Because of the close ties of this technology to the society it supports that society becomes protective of the technology and its place of use and develops some assessment capability for determining threats to that technology and hence to their



*3b. regulatory response
3c. technological response
4. regulatory threat

Figure 1 THREATS AND RESPONSES FROM TECHNOLOGIES ASSOCIATED WITH DIFFERENT SOCIAL VALUES way-of-life and overall environmental values. Should any external threats be perceived or actually felt responses are generated in either a direct threat to the offending parties, a direct approach to the regulator concerned, an alternative technology for meeting the problem which, in turn, must be approved by that regulator, creation of a new regulator or some combination of all of these responses.

In the case of society's energy values some existing technology can be creating some set of problems or other inadequacies in fulfilling these needs and values. Given the perceived energy demands or resolution of these problems a new technology is generated through research and development of some pilot plant. Over time this technology is approved by some regulator and becomes adopted as the accepted new way of meeting society's energy needs. At this point a new social system develops in the energy bureaucracy as well as in supporting institutions to maintain the status of this adopted technology. The protection of this technological system is fostered through an assessment capability that determines whether threats, either external or internal, exist and generates a set of responses. If the threat is seen as external from another social-technological system based on, say, environmental values then responses are formed to counter this threat. The regulatory body that approves and authorizes technologies of, say, energy is often the focal point for such conflict. Responses can be external to the regulator, a direct attempt to fix the existing regulatory stance or provide a new technological capability for meeting regulatory shifts. Internal threats are generated within the energy system from some competing energy source.

The kinds of responses that affect the energy system and its regulator from those with, say, environmental values, include some of the following:

- change society's energy values (external threat);
- ban the new technology (external and regulatory threat);
- pursue adoption of an alternative technology (external technological threat);
- attempt to create another or change the regulator (external threat to regulatory and energy system);
- create new standards for technology (technological threat);
- change existing standards for approval and acceptance (technological threat).

Each of these responses are often attempted in concert; however, the thrust of this paper is toward standard-setting and it is interesting to note the role of standards in such responses above. Either setting new standards or changing existing standards can provide major channels for those opposed to accepted energy values. This acceptance, of course, includes those in the energy system as well as those regulating it via approval of the new energy technology.

Those in the energy system also respond to such threats; however, these threats are normally confined to the regulator with whom they have evolved a rapport. Such responses include:

- counter environmental arguments directly to regulator
- ask for standards to be created
- ask for a tightening of existing standards
- generate newer technology internally.

Again standards can play a role in the responses of those supporting a, say, large energy system. Again the responses are either setting standards where none existed or tightening existing standards as a means to forestall further actions from environmentalists.

THE CONCEPT OF ACTORS

Thus from Figure 1 and the above discussion it is readily seen that a system of protection does develop around a set technology. This protective system includes a wide range of actors that have essentially the same set of values or core beliefs in the technology as a means to achieving their economic and personal goals. Such actors for an energy system would normally include:

- industry characterizing the energy source;
- supporting industry, including contractors and suppliers;
- supporting unions, workers and company towns;
- supporting research and development institutes;
- supporting journalists and publications;
- government regulator;
- allied government units, including energy, finance, and industry;
- allied international governments and agencies;
- affiliated financiers;
- infrastructure suppliers and contractors;
- public utilities;
- consumer associations, including industry, commerce and home;
- political representatives associated with the energy source;
- local govenments in towns centred around the energy source.

As can be readily seen this actor array includes a wide spectrum of support for an energy system. This array, while apparently unlinked in any organizational fashion, would center around a set energy technology that meets their prevailing energy and economic values and needs. The existence of some external threat to the energy system is the basis for bringing together these seemingly disparate parts into a cohesive system for protecting the technology on which all depend for their identity and livelihood.

One important factor running through the above is the concept of actors and associated values as demonstrated via the degree of favorability toward the technology and the degree of involvement with it. Figure 2 attempts to show the array of possible actor groups which influence the outcome of a, say, petroleum development programme [5].

HIGH - CENTRAL ACTORS	CORE ACTORS	INDEPENDENT CENTRAL ACTORS	STRONG RIVALS AND ADVERSARIES
↑ INVOLVEMENT IN PETROLEUM DEVELOPMENT PROGRAMME ↓	ALLIED SUPPORTING ACTORS	MIDDLE RANGE ACTORS • BENEFICIARIES • SUFFERERS • CONDITIONAL	TRANSITIONAL RIVALS AND ADVERSARIES
LOW - PERIPHERAL ACTORS	EXOGENOUS SUPPORTING ACTORS	EXOGENOUS INDEPENDENT ACTORS	EXOGENOUS RIVALS AND ADVERSARIES

HIGH ← FAVORABILITY TOWARDS PETROLEUM DEVELOPMENT PROGRAMME → LOW

Figure 2 ACTOR CLASSIFICATION FRAMEWORK

The left-hand column represents actors with varying levels of involvement who are all favorably disposed to the development programme. As the degree of involvement decreases the specificity of the supportive relationship will likely change from intimate knowledge of the development activities on the part of core actors to more generalized supportive value orientations on the part of "supporting exogenous actors".

The middle column represents a "mixed" attitude or value orientation. Actor types would include:

1. Neutral actors.
2. Should-be actors.
3. Independent actors who theoretically are without *a priori* value positions. Such actors may conduct "objective" analyses. Regulatory agencies whose concern is the "national interest" would be classified as independent actors.
4. Conditional actors who depend on decisions of other actors (as well as own decisions) these actors may be either favorably or unfavorably disposed to a development programme.

The right-hand column of the framework consists of those actors opposed to the development programme: the adversaries and rivals. Reasons for opposition will vary among such actors and be related to their own particular objectives and the situation or context in which they are located. Rivals and adversaries demonstrate varying degrees of involvement. Even those peripheral to the policy system may significantly affect that system including the petroleum programme. Clearly, decisions in the international sphere such as the OPEC oil price increases have had an impact on petroleum development in the North Sea.

In order to clarify the actor classification framework a description of each actor-type follows.

- Core Actors
  This group has continuous and intensive involvement in the technological development programme. Though other actor-types may make fundamental decisions it is usually the core actors who initiate a programme via one or more fundamental decisions. Core actors are usually fewer in number than other actor-types though core actors may change as the development programme evolves through major phases (e.g., exploration, production, transportation, etc.).

- Allied Supporting Actors
  These actors are characterized by a positive or favorable orientation to the development programme. Their activities enhance the development programme. Supplying of goods and services, provision of infrastructures, enabling decisions, passage of legislation, and so on, are examples.

- Independent Central Actors
  Actors of this type have a degree of independence or autonomy from both the proponents and adversaries of a given development programme. Their autonomy may be either constitutionally or legally-based or a function of an "objective" information position, i.e. performing their own research, information-gathering and interpretation.

- Middle Range Actors
  This actor type has moderate involvement in the development programme and may have favorable, unfavorable or neutral attitudes to the development. An actor is classified middle range for several reasons:
  1. chooses only moderate involvement,
  2. has expertise peripheral to programme,
  3. has limited legal basis for involvement,
  4. lacks information especially from other key actors, thus position not sufficiently well developed to qualify as another actor-type, or
  5. is waiting for decisions by others to determine or declare position.

- Transitional Rivals and Adversaries
  This group of actors consists of those who, for reasons of expertise, power, resources and information, have only moderate involvement and are declared rivals or adversaries of the technological development programme. Not only may actors be emerging as rivals and adversaries *but* also, given a fundamental decision against the prevailing programme, former "core" or "allied supporting actors" may shift to rival or adversary status, while those who were formerly rivals and adversaries assume "core" or "allied supporting actor" status.

- Strong Rivals and Adversaries
  Such actors are characterized for the most part by having developed viable alternate technological programmes. To the extent that two or more development programmes are recognized (one of which is that of the core actors) and are similarly feasible, then strong rivals will exist. Strong adversaries might include political organizations who may be central to the development programme in terms of power but ideologically opposed to the particular technological-economic programme mix.

- Exogenous Rivals and Adversaries
  This group of actors is outside the technology policy system, certainly in its day-to-day, week-to-week functioning. Exclusion may be on a geopolitical basis (e.g., another country). They are opposed to the prevailing development programme, and they may support an alternative programme which may or may not differ technologically.

- Exogenous Independent Actors
  These actors are seen as exogenous to the policy system, usually for geopolitical reasons. Like "independent central actors" they have an independent or autonomous role due to constitutional or legal factors.

- Exogenous Supporting Actors
  1. Those actors who have definite links to "allied supporting" and "core" actors and who though geopolitically distinct from members of the policy system may indirectly and significantly control their actions (e.g., multinational corporations).
  2. Those actors who are characterized both as marginal to the development programme and supporting it. Usually they can only be identified through solicitation or indirect representation of their views (e.g., other countries with similar development programmes).

The actors classified as above provide a systems perspective of the basic behavioral structure between the energy system and some other system, for example, environment as shown in Figure 1. This actor linkage and interplay, given the descriptions of each position above, can be manifested in a variety of ways and strategies.

ENERGY SYSTEM ACTORS

It is readily seen from Figure 2 that the interface of the energy system with some other system, say, environment is complex, representing numerous actors and decision-makers with diverse, indirect and subtle positions, often containing conflicting objectives. Therefore, when considering individual measures such as environmental standards, many of these actors can be regrouped to demonstrate only the essential interactions and interests.

Figure 2 ends at the point of attempting to display an energy technology threat via some regulatory process. Figure 3 is an attempt at showing the actor array involved in the standard-setting process. This figure places major actor groups together into a system that links the groups expected to participate in any regulation of a technological system. Figure 2 has been useful in structuring the spectrum of actors one could expect in such a regulatory process and provides the basis for those actors in Figure 3.

The approach suggested by Figure 3 can be seen as basic to some of the cases reported herein. An energy system defined on the basis of actors shows the minimal actor configuration required for an adequate solution to some threat or other problem of interest. Each of these actor sub-systems comes into the energy system on the basis of different values and for different reasons, such as:

- development actors:
  value - profit or service potential
  reason in system - generation of energy via approved technology

- regulatory actors:
  value - assimilate opposing demands and approve technology
  reason in system - legislative/administrative authority

- environmental expert actors:
  value - professional and personal interest
  reason in system - recognition of energy impacts

- impactees:
  value - preservation of environment/livelihood
  reason in system - dissatisfaction with potential outcome

- exogenous actors:
  value - preservation of environment/energy systems
  reason in system - treaty, international influence.

Each actor sub-system within the energy system can be seen as a set of individual decision-making units that goes through some decision-making process in its particular decision field. Placed together these units provide a systems perspective. These decision-making units, such as in the regulatory sub-system, must define both their internal objectives and their alternatives. Their own self-interest is usually predominant even if such objectives are initially determined by an exogenous unit, i.e., a legislative body. The alternatives perceived by the unit are those that mesh with their objectives. In determining alternatives the unit must assess information and external actions and events that define the value relevant dimensions of its alternatives (and even of its objectives). Given its assessment of its information and external constitutencies, the unit must then evaluate the probable consequences of its potential decisions against the consequences emanating from that decision for its objectives. Part of the consequences include the value of the decision to the other decision-making units or actor sub-systems in the overall energy system. Finally, the unit selects the decision seen as most "correct" given the above steps. This procedure is readily recognizable as the rational model of decision-making. The rational approach can be embedded in each actor sub-system of the energy system to some perceived threat but it does not follow that a search for a completely rational solution will occur in the overall system, much less between systems based on different technologies and social values[6].

ACTORS AND VALUES

In principle, there should be a unique correspondence between the values describing the position of a single actor category which can then be juxtaposed against other actors. While Figures 2 and 3 suggest the kinds of values that each actor grouping may hold, the values found are often very general, overlapping and interrelated. In a complex and large-scale energy programme such as found in the North Sea petroleum effort, few actors were found with unique or single-dimensioned values. Indeed, many central actors have attempted to become so broad in outlook that the decision-making system can appear even more diffuse. What emerges as a central issue is the defining of the boundaries of the decision problem and the means for addressing separate policy issues, such as standard-setting.

For example, some of the kinds of values one could expect to be included in the North Sea energy system would include those listed in Table 1. These values are quite broad and must be re-arranged via such devices as a goal tree in order to understand their hierarchy and subsequent embedding.

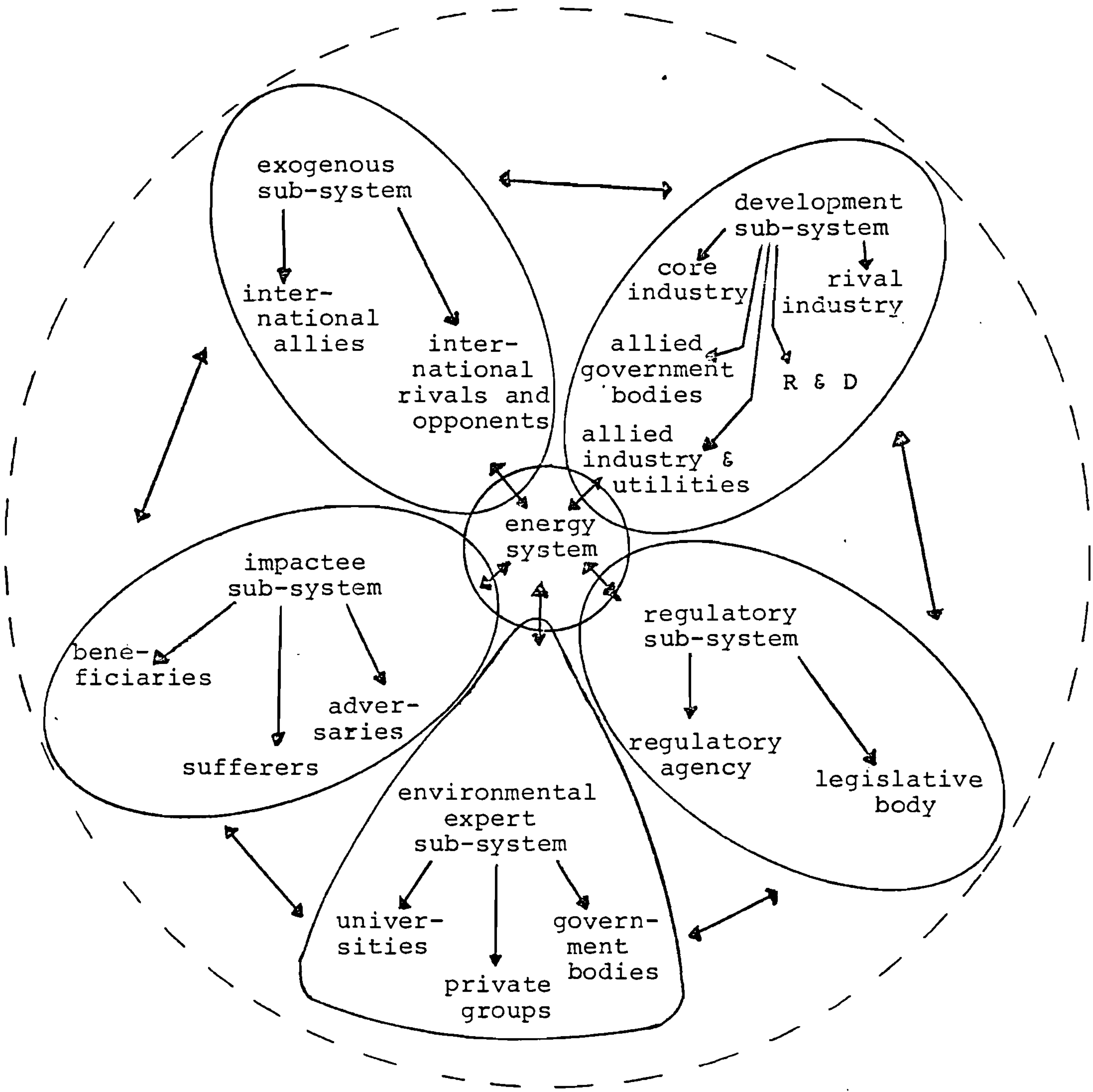


Figure 3 THE ACTOR SUB-GROUPS INVOLVED IN AN ENERGY SYSTEM

Table 1 NORTH SEA PETROLEUM DEVELOPMENT ACTORS AND VALUES

UNITED KINGDOM		NORWAY	
ACTORS	VALUES	ACTORS	VALUES
A. CORE aa. oil industry ab. BNOC ac. Dept. Energy (DOEn.) ad. British Gas Corp.	• cash flow • government shares • public image • rapid development • reduce oil dependence	A. CORE aa. oil industry ab. Statoil ac. Oil Directorate ad. Min. Industry	• stable industry • government participation • public image
B. ALLIED ba. Dept. Trade (DOT) bb. Cabinet bc. Treasury bd. Scottish Office be. Scottish Dev. Dept. (SDD) bf. Dept. Environment (CUEP) bg. local industry & unions bh. suppliers bi. National Coal Board	• balance of payments • induced growth • Scottish development • enhance economic image • additional revenues	B. ALLIED ba. Dept. Finance bb. Maritime Directorate	• stable economy • accommodate impacts • capture all benefits
C. INDEPENDENT ca. Nature Conserv. Coun. (NCC) cb. Natural Environmental Research Council (NERC) cc. Min. of Agr., Fish & Food (MAFF) cd. government labs. ce. river authorities cf. local governments cg. Countryside Comm.Scot.(CCS)	• environmental quality • public health	C. INDEPENDENT ca. Min. Environment cb. Min. Fisheries cc. Inst. Marine Research	• retention of old economic activities • environmental quality • public health
D. MIDDLE-RANGE da. local conservationalists db. local environmentalists dc. university professor dd. local business de. Dept. of Agr. & Fish (DAF)	• preservation of old opportunities • local amenity • resource over time • loss of business	D. MIDDLE-RANGE da. Nor. Soc. for Cons. of Nature db. Council for Env. Studies dc. Nor. Board of Fisheries	• preservation of old opportunities
E. RIVALS AND ADVERSARIES ea. Scottish Nationalist Party eb. Friends of Earth (FOE) ec. Sierra Club ed. fishermen ee. nuclear energy ef. church	• Scottish independence • environmental quality • maintenance of fishing patterns • energy balance	E. RIVALS & ADVERSARIES ea. university professor	• uncompensated changes in economy and biota • resource-wise balanced growth
F. EXOGENOUS fa. European Economic Com. (EEC) fb. OECD fc. OPEC fd. multinational oil corps. fe. Norway	• oil sharing • oil reserves	F. EXOGENOUS fa. Nordic community fb. United Kingdom	• oil sharing • oil reserves

One clear observation is the deepening of values over time as the particular energy programme proceeds. Determining a stable development path for the economy as well as the environment in such a complex array of value positions can prove difficult. The time horizons of such values cannot be wholly relied upon for guidance of policy responses as both short and long-term contradictory objectives and expectations may exist within the same value set such as short-term local public revenues from oil companies versus long-term slanting and dependence of such local economies.

Interplay of actors, and therefore interplay of controls such as standards, results in indirect relations. For example, ecology itself is not a value in which companies would be directly interested, but environmental goals influencing company capital and operating costs are of interest (which in turn are of no direct importance to environmental bodies). Translating actors' values into environmental standards cannot be done directly without the intermediate steps of identifying products and impacts associated with the various stages of the proposed development programme. Next, these products and impacts must be arrayed against the actors involved to determine their degree of interest and influence in changing the outcomes. Finally, the mechanisms for change or acceptance must be seen between all such actors in both policy systems.

The important point here is to demonstrate the necessity for a comprehensive systems perspective for understanding the decision-making context in which each environmental measure is embedded. The criteria of actor comprehensiveness as in Figures 2 and 3 can be used in deriving this framework. Out of such a synthesis the appropriate boundaries for an analytical approach can be derived for observing the essential elements of the problem and structuring it for relevant analysis in its systems context. Indeed, the environmental responses of standard-setting, monitoring, planning, etc., emanate from the synthesis just described.

ACTORS AND STRATEGIES

Actors tend to become involved in a specific site development issue if they can expect to have some significant impact on the outcome. The different levels of involvement and favorability set forth the dimensions of the decision to invest effort into attempting to support or shift the potential outcomes via certain strategies. Those actors with local ties to the site tend to focus on those avenues giving weight to the local point of view such as the legislative representatives, while actors having direct ties with the technology emphasize institutions weighted toward technology such as the regulator or a specialized legislative committee or agency.

Establishing the focus of the energy system is in itself a strategic consideration. Energy technology-associated actors are often served by a narrow problem or issue definition which limits the scope of information necessary, deflects attention from broader issues, precludes participation of certain actors and obscures connections with alternative means. Environmental-oriented actors tend to define the issues more broadly to encompass other social and economic goals that are in conflict with the goals which the offending technology serves and otherwise open up the decision-making process to other actors and related information. Purely site-oriented actors focus on their economic livelihood and traditional lifestyle and tend to use the traditional political process to impact on the regulator.

Consent-building and conflict containment represent two major strategems used in responding to threats. Consent among actors involves command relationships where direct control is exercised as well as bargaining and persuasive modes of creating agreement where no one actor has direct control. Whenever the effective power base of an actor is insufficient to achieve their desired level of protection and influence attempts are made to create a coalition with allied actors. Such coalitions would affect finance, information and could even modify the technology itself through certain accommodations to the other actors forming the coalition. Changing the image of the technology or adding other components such as an environmental buffer zone are also used to generate additional support. Creating or exploiting a crisis is also used to build majority support whether from an energy technology or an environmental point of view.

Containing conflict can also be used as a strategy should little chance of an agreement among actors exist. Those actors favoring a technological system work with their regulator to routinize the acceptance process thereby rendering somewhat automatic decisions. Technical "tests" are used as a means to ensure an orderly development process and to avoid conflict. Ultimate goals are avoided in such a process. Should significant opposition occur then additional components can be used to generate acceptance or the threat of loss of the project altogether to obtain greater local support among the beneficiaries. Actors not tied to a local base of power, but who depend upon an amorphous general public for support, can be isolated and excluded by claims of irrelevance and their unwillingness to be accommodated through the normal regulatory process.

ENERGY SYSTEM LINKAGES

Figures 4 and 5 show the interface between the energy and environment systems in the UK and Norway. Both standard-setting systems vary considerably even though their final standards are similar. In the UK the energy system itself has final authority for setting standards for chronic offshore oil discharges while in Norway the environment system has such authority. Thus,

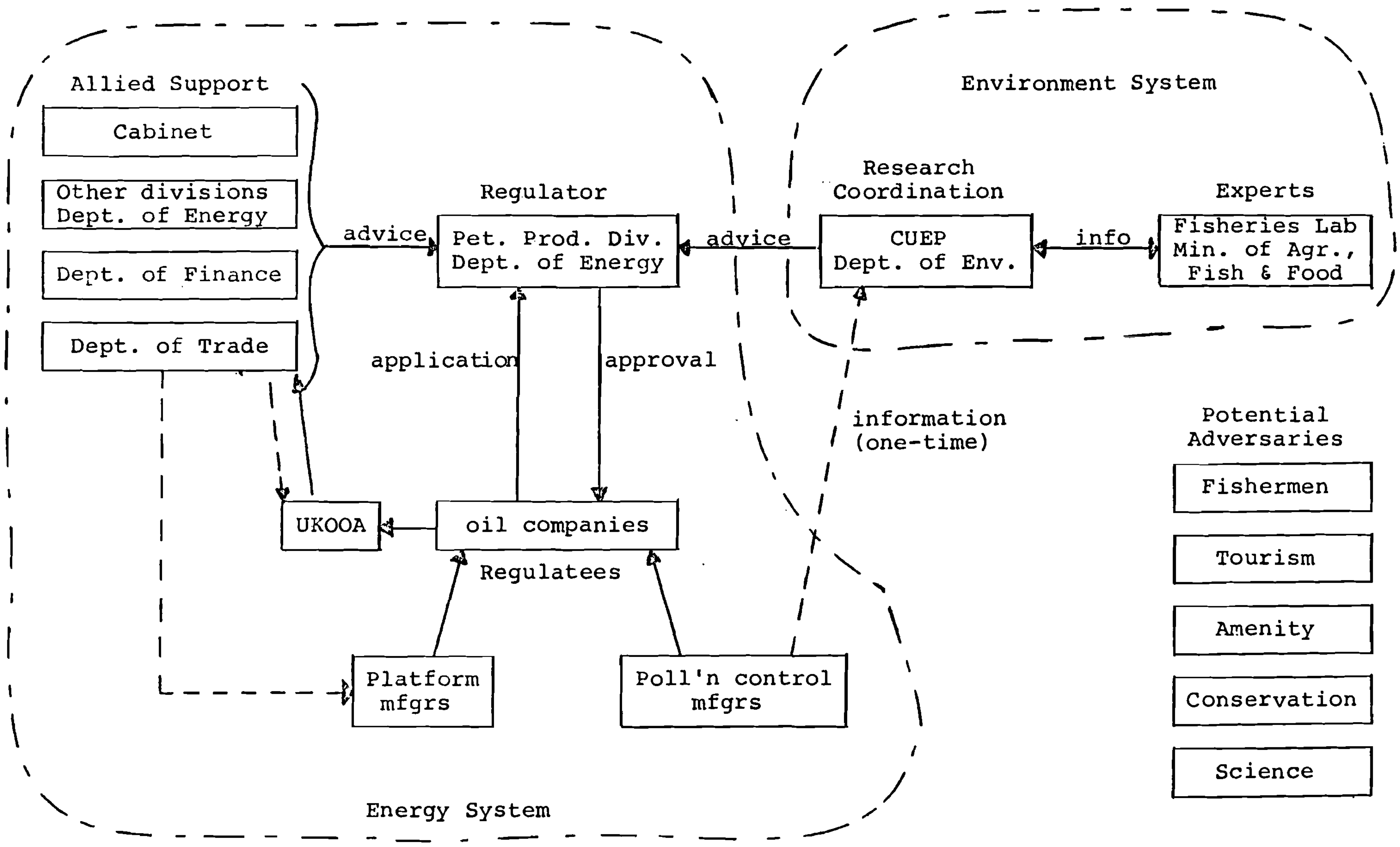


Figure 4 AN ASSESSMENT OF THE ENERGY-ENVIRONMENT INTERFACE IN THE UK FOR SETTING STANDARDS FOR OFFSHORE CHRONIC OIL DISCHARGES

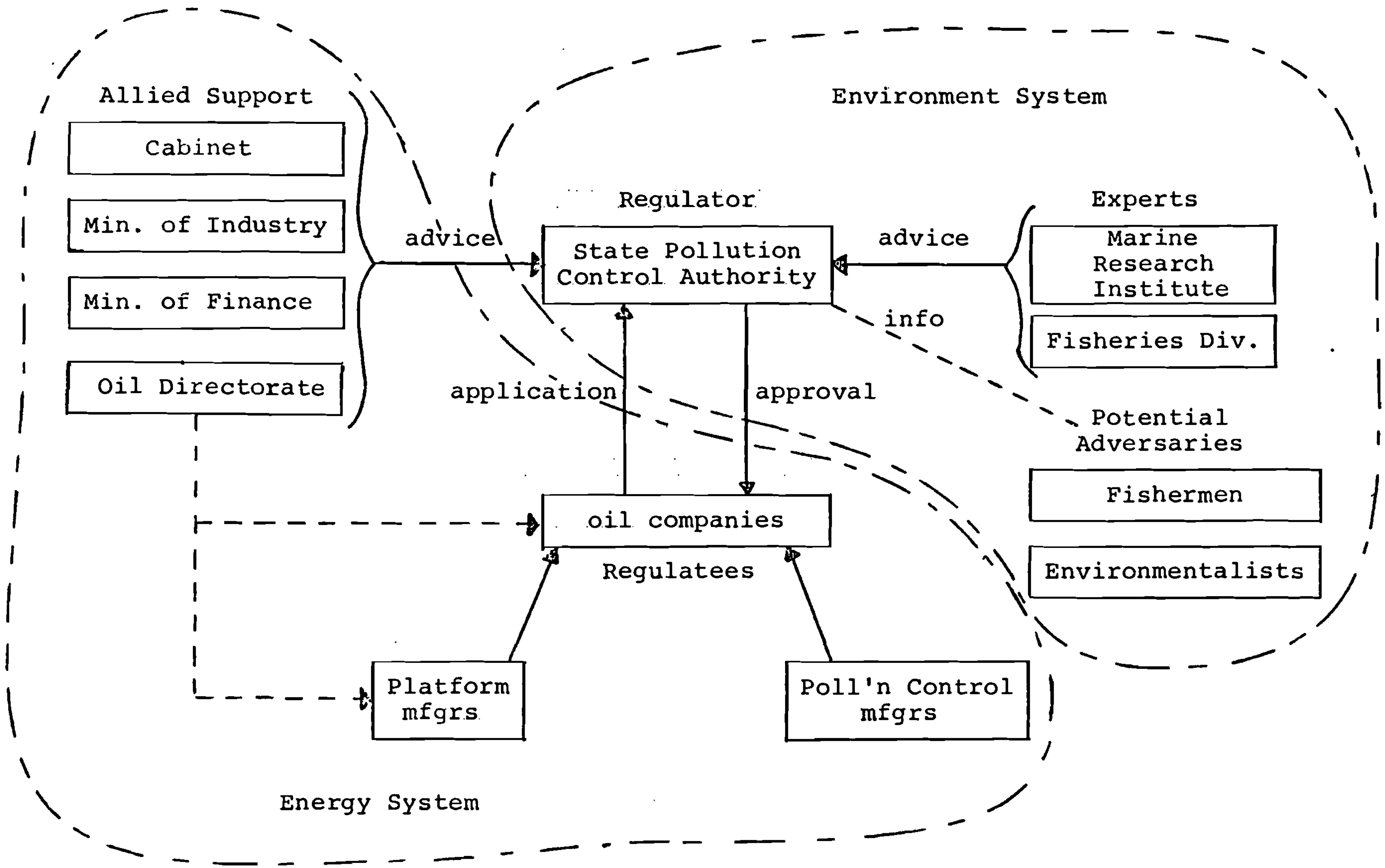


Figure 5 AN ASSESSMENT OF THE ENERGY-ENVIRONMENT INTERFACE IN NORWAY FOR SETTING STANDARDS FOR OFFSHORE CHRONIC OIL DISCHARGES

greater pressures of an implicit nature could be brought to bear within the energy system to keep standards from inhibiting the pace of offshore oil development in the UK. The pressures here can be thought of as "least possible change" pressures that provide a basis for coping with pollution control demands by outsiders or those in the environment system. Within the UK energy system, informal information channels operate to meet and diffuse changes or threats of change that would go beyond pollution control standards or generate standards beyond "practicable" levels. Because the regulator in the UK is part of the energy system and indeed is part of the very Division interested in increased petroleum production it is interesting to find standards similar to those in Norway. The one important leverage in the UK is the individual applications for exemptions to the 1971 act prohibiting all oil discharges offshore. No exemptions have failed to be approved, however, and sanctions against oil companies are not prevalent.

The environment system in the UK is on a much smaller scale than the energy system, and does not include potential environmental supporters and allies. Thus, these same external entities, although often diffused, become potential adversaries to both of the energy and environment systems. No formal links exist to connect these potential actors and impactees to either system. Therefore, since regular avenues are closed to them, other channels must be created by them, such as confrontation or other non-routine interventions in either system.

The environment system has only one direct channel to the energy system, the advisory role of the Pollution Unit. Since this unit has no executive authority it must rely on persuasion in an informal working relationship with the energy stem to influence its standard-setting on a case-by-case basis. The only other links to the energy system from the environment system were the information provided by pollution control equipment manufacturers and the seminar for energy system participants held by the Pollution Unit on such equipment alternatives. Both of these links were conducted on a one-time basis.

In Norway, the energy system also over-balances the environment system, but environmental values are more deeply embedded in the overall decision-making process that governs both systems. The greatest differences in Norway include the regulator being in the environment system, as well as having the authority and expertise matched within the same agency. Supplementing its resident expertise, the Pollution Authority maintains ties to other government experts, as well as to potential adversaries, the fishing and environmental organizations. Part of the reasons for doing so is the uncertainty problem and the need to have allies to offset the larger organized resources allied to the energy system. Thus, a greater spectrum of expertise is available to the State Pollution Control Authority on a regular basis.

Missing links in both the UK and Norway energy-environment systems would seem to be the private sector both within the energy system and between the two systems. In the energy system the largest constraint to having pollution control equipment installed on offshore platforms is the space constraint on such platforms. Pollution control is seen by oil companies and platform manufacturers as an "add-on" rather than as an integral part of the platform design and operation itself. Thus, it would seem important to establish a link between platform and pollution control equipment manufacturers. This link should serve as a means for integrating designs and operational capabilities thereby reducing constraints and costs of pollution control.

The other missing link would appear to be between the private pollution control equipment manufacturers and the environment system. Regular links would provide a better basis for ascertaining availability and capability of such equipment by those responsible for advising the regulator. In addition, environmental research findings could be given to the pollution control equipment manufacturers for redesigning performance capabilities. Possible subsidies for development of new approaches in the private sector could also be fostered.

Given the two systems in their present configurations the direct beneficiaries and "sufferers" (impactees) include only the oil companies and manufacturers of pollution control equipment. Regulators' decisions only directly affect these two groups. Fishermen and environmentalists have less direct and, indeed, tenuous impacts from decisions by the regulators. The platform decision is made entirely within the energy system with no environmental inputs as to location or design. Only the add-on technology of pollution control is the issue to the regulator, oil company and manufacturer.

WIDENING THE STANDARD-SETTING PROCESS

Table 2 presents a summary of many of the facets involved in the process of setting standards for chronic oil discharges from offshore platforms. Since much of the table is self-explanatory, only certain highlights will be noted here. The table suggests categories of information capsulizing key elements of importance in the standard-setting process. Such information is grouped under actors, choices, associated choices and hence actors, choice criterion, uncertainties and conflicts.

It is interesting to see that the regulator, developer and environmental experts have roles that exceed those of the impactees, while exogenous actors provide general constraints to the national systems. The impactees who have the greatest need for standards and are part of the reason for the existence of such standards have the least role in the setting of standards.

Table 2 SUMMARY OF OFFSHORE OIL DISCHARGE STANDARD-SETTING PROCESS

	REGULATORS	DEVELOPERS	IMPACTEES	EXPERTS	EXOGENOUS
Key actors	UK: PPD, DOEn. Nor.: SPCA, Min. of Env.	- oil comps. - poll'n control eq. mfgrs.	- fishing industry	- poll'n control research - fisheries research - marine research	- EEC - others
Key areas of choice	- set standards - meet production goals	- implementation of standards - eq. alternatives to meet standard	- effects of standards on catch - change fishing pattern	- standards alternatives - how monitor effect of standard	- set constraints on national standards
Associated choices	- condition of development (exemption) - monitoring eq. - poll'n control eq.	- purchase of eq. - monitoring eq. - R & D	- monitor catch	- costs of standards - tests of eq. - monitor ambient env. & samples of fish	- effects of standards on trade & development - quality of North Sea env.
Associated actors	- Energy - Industry - Finance	- other oil comps. - other eq. mfgrs.	- other fishing nations	- food, fisheries - other researchers	- non-EEC countries
Choice criteria	- acceptability of standard to comps. - government image & obligations	- cost of implementing standard - cost of maintaining relationships - R & D costs	- sales volume - way of life	- public reactions - eq. capabilities - research capabilities & resources	- EEC agreements - other nations agreements
Uncertainties	- standard's value to env. - standards impact on comp. - poll'n ctl. eq. availability	- basis of standards - future government standards	- future market value & volume	- claims of mfgrs of poll'n ctl. eq. - effects on env. - non-technological means for poll'n control	- effects on trade - effects on env.
Conflicts	- internal: production vs. env. - external: informal, closed vs. formal, open decision process - requests vs. requirements	- cost vs. need - standards vs. guidelines - maintain relations vs. fight standards	- access to decision process	- researched advice vs. best guess - gaining research funding support - env. perspective vs. technology perspective	- international vs. national emphasis

From a systems perspective, the standard-setting problem does not appear to be cast into the systems approach in either country. The necessary perspective centers around the responses by the energy and environment systems in a standard-setting process. It is here that such criteria as comprehensiveness and integrativeness can come into play. Comprehensiveness can be defined in two ways:

- a comprehensive set of actor groups or sub-systems included in the standard-setting process, and
- the full range (comprehensive) of environmental management activities necessary to regulate the development activities, including research, planning, regulation, management and monitoring in support of standard setting.

Integrativeness can also be equally defined in two ways:

- an integrated set of relationships between actor groups or sub-systems, and
- an integrated communication pattern giving similar weighting to information generated.

To approach these two criteria, the energy and environment systems would be expected to respond jointly to the challenge of environmental management through the creation of the proper means for achieving adequate resolution to environmental problems generated by energy activities.

"Muddling through" still characterizes some of the responses of industry and government, such as industry providing no alternatives to a proposal or government merely reacting to industry initiatives both of which only reinforce traditional patterns of assessment and response. A common pattern is to drift into joint solutions to environmental problems between industry and government where circumstance and expedience require or justify action even before the inherent environmental complexities are made clear. The wide array of actors and potential actors to become involved in energy-environment responses appear to play a lesser role than would seem warranted from the lack of environmental information and alternatives known to exist. Drifting into joint solutions among core actors with wide discretion and informal relationships can distort the broader process of decision-making where representative bodies should play a major role. Because the government (majority party) can define the terms of its relationship with outside interests, actors and pressure groups, there is little requirement for these outside actor groups to be concerned with or go through the Parliament. Such decision strategies vis-a-vis other actors raise issues that affect the decision-making or standard-setting system itself.

The systems perspective here emphasize comprehensive and integrated responses to environmental management. It is clear that such a response pattern would be quite easy to set aside since the results of a systems approach would not show in the immediate future, except for accident responses which are often soon forgotten by the public. While core actors do dominate the decision-making system and do attempt to influence other parts of the system to their viewpoint it is clear that other actors are becoming more vocal. The actual persuasiveness of other actors is still open to consideration. Nevertheless, as core actors, both the oil industry and the requisite government departments have begun to recognize the necessity of expanding their information network, pressure points and feedback patterns.

In order for standards to be set in a conflict situation, certain inter-system linkages should be developed. Some of the linkages one could expect include:

- integrated policy objectives and planning mechanisms: the integration of environmental quality objectives and planning precepts into the energy objectives and planning approaches at the highest level;
- multi-disciplinary involvement: the involvement of relevant, experienced professionals from a variety of disciplines for generating a broader response spectrum;
- public participation: the participation of a broad spectrum of publics representing the relevant external value dimensions of each system;
- systems perspective: the perspective derived from a systems analysis and synthesis for denoting the full range of impacts, actors and responses;
- reticulist role [7]: the creation of a special inter-system role to provide a communications link between policy systems for joint exploration of inter-linked responses;
- unified monitoring and communications system: the development of a comprehensive and unified system of data collection on the baseline, development impacts, their effects and feedback to publics and actors in the relevant policy systems.

Such inter-system links as these constitute examples of how the energy-environment interface can be bridged. Taken together these examples provide a technology assessment system capability for obtaining the broadest possible assessment of the inter-system implications of a major technologically oriented development programme [8]. These inter-system linkages can occur through and be integrated into most of the normal operations associated with any major technology programme such as finance, personnel, accounting, etc.

One key element that combines both a comprehensive view and an integrative approach is that of the reticulist role suggested by Friend, Power and Yewlett as noted above. The reticulist becomes the system assessor of some major technological development system. This role literally means networking judgements which are required in any policy system or decision network that occupies the interface between two contiguous policy systems such as development and environment.

The environmental management system in such a vast and unknown area as, say, the North Sea is characterised by situations which call for a joint energy-environment systems response. The roles generated by the core actors of government and industry have worked well to implement the development goals of these actors, namely the maintenance of a rapid development pace and increasing scale of development. However, this role has worked only partially well in the overall environmental area, despite efforts by the industry and government. The fact that impactees and outside experts have a low profile in decision-making is shown through the mismatch of roles. Such issues as the above suggest that the decision-making system itself may not only be lacking in such respects as including impactees from offshore oil development and non-government affiliated experts, but that it is not responding comprehensively nore in an integrated way. A comprehensive technology assessment system would provide for the inclusion of all actor groups arfected by offshore petroleum development [9]. Only those interests will be considered in any assessment when all interested or affected parties are directly involved in the assessment process.

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SETTING STANDARDS FOR CHRONIC OIL DISCHARGES IN THE NORTH SEA*

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Summary

Public attention has frequently been directed to the large accidental oil spills connected with tanker groundings or break-ups and platform blowouts. The operational oil discharges connected with offshore production platforms could, however, have equally severe effects in the offshore environment because the pollution occurs day after day as long as the activities that produce it continue to operate. This study is concerned with such discharges in the North Sea.

The study was focussed on the central actors involved in the standard-setting process for operational or chronic discharges. Each key actor was interviewed in depth to determine their perceptions of the problem, the role of other actors, the alternatives considered and the interactions among actors. This decision-making complex thus includes a variety of decision-making units with varying abilities to influence the outcomes of the standards. Regulator, developer and environmental expert actors formed the basic focus for this effort.

The objectives of the regulators and developers are described along with the alternative regulations, treatments and uses of the environment that are considered. Then a comparison and evaluation is made of the decision process by which Norwegian and U.K. regulators arrived at standards.

Major findings include the near similarity of the standards set even though the approaches to such standards are perceived to be quite different in each country. Another finding is that entrepreneurial endeavor for oil discharge treatment equipment is a key parameter in deciding on standards since both countries must rely on such effort for what is available. Lack of information on the effects of oil in the sea is another finding along with a minimal role for environmental quality factors in setting standards. Such findings make it difficult to trace trade-offs in the selection process for standard-setting. Finally, some suggestions for improvements in this process are noted.

Contents

Setting Standards for
Chronic Oil Discharges in the North Sea

1. AN INTRODUCTION TO THE PROBLEM OF OIL DISCHARGE REGULATION IN THE NORTH SEA

Public attention and environmental concern with North Sea oil operations have traditionally centered on tanker accidents and oil blowouts from drilling and production platforms. This emphasis on accidental oil spills is quite understandable considering the dramatic nature of such events, the publicity they receive, and the damages they can do to the environment. Oil slicks on amenity beaches and disabled sea gulls come to mind when one thinks of accidental oil spills.

Meanwhile, another type of oil pollution has received relatively little public attention: pollution through chronic (operational) oil discharges from ships and offshore production platforms. Washing and ballast water are the main sources of chronic oil pollution from ships. Production water from oil-water separation processes and displacement water from storage tanks are routinely discharged from production platforms. Although accidental spills are more visible and can have a large and immediate local impact, chronic discharges with low oil concentration levels continue over the years and may have long term environmental effects.

The total amount of oil entering the sea through chronic discharges is by no means small in comparison to accidental discharges. Table 1 presents estimates of the United States Academy of Sciences for the early 1980s. [1]

Table 1

Estimates of world wide oil pollution in the seas resulting from from offshore oil activities
(tons per year; source: US Academy of Sciences, 1975)

	ACCIDENTAL	CHRONIC
TANKERS	150 000	200 000
PLATFORMS/ PIPELINES	150 000	50 000

These discharges are part of a total of 4.57 million tons of discharges from all sources including natural seeps, river runoff, etc. The figure of 200 000 tons for chronic discharges from tankers cited here refers only to ballast water, and it is an optimistic estimate based on a strict enforcement of recent international tanker regulations. According to the US Academy of Sciences report chronic discharges from tankers are presently more in the area of 1 million tons per year.

How do these world-wide figures translate into the North Sea context? Estimates of the increase in oil pollution produced by North Sea oil operations vary widely. An early report by the Norwegian Ministry of Finance [2] stated a figure of 12 000 tons of total oil discharges per 50 million tons of oil produced, shipped, and stored. At present peak production estimates of about 200 million tons for 1981, this would amount to approximately 48 000 tons of oil discharges per year, including accidental and chronic discharges.

The Central Unit on Environmental Pollution (CUEP) of the UK made some estimates of chronic oil discharges from UK operations including the Norwegian Ekofisk field. [3] Assuming rather severe treatment levels for oily water they calculated that oil operations would result in approximately 2000 tons of chronic discharges per year.

Based on MIT statistics of accidental spill rates and amounts, US Academy of Sciences statistics of chronic spill rates, and North Sea statistics cited by two reports of the CUEP, [4,5], estimates for chronic and accidental oil pollution resulting from North Sea oil development in 1981 were made by the authors. These are summarized in Table 2.

Table 2

Estimated increases in oil pollution resulting from North Sea oil operations (tons per year)

	ACCIDENTAL	CHRONIC
TANKERS	1300	2400
PLATFORMS/ PIPELINES	5000	2200

Chronic oil discharges thus emerge as an equally severe problem as accidental discharges if one looks at total discharge figures.

Some regulatory measures to reduce chronic oil discharges from ships and platforms had already been taken by the UK and Norway before North Sea oil production began. Both countries follow the tanker regulations set forth by the Intergovernmental Maritime Consultative Organization (IMCO). [6] Oil discharges from platforms were strictly forbidden by the Prevention of Oil Pollution Act of 1971 (United Kingdom). [7] In practice, this regulation became obsolete with growing oil development in the North Sea. The initial solution to the dilemma that was posed by the Act of 1971 was to issue exemptions to offshore operators under the Petroleum and Pipelines Act of 1975. [8] This Act allows an exemption to be issued to operators provided they use "best practicable" means to reduce the oil content in discharged water. But neither the strict prohibition of 1971 nor the loose exemption rules of 1975 could be the final word about chronic oil discharge regulations. More precise definitions of "best practicable" means were needed.

With respect to offshore platform discharges, Norway faced a similar situation as the UK. Norwegian pollution control policy was to require operators to use "best applicable" means to reduce the oil content in discharge water. But again there was no firm rule or procedure by which the regulator could establish or communicate what was meant by "best applicable" means.

In both countries the main problem was to operationalize their respective regulatory definitions. Both countries are taking the route of setting standards on oil concentration, although levels and application rules differ in each.

The following sections present an analysis of the processes by which Norwegian and UK set standards on chronic oil discharges from offshore production platforms, and how they apply these standards in the day-to-day process of handling individual applications from oil developers. The information on which this analysis is based was collected during two field studies in Norway and the UK during which researchers, governmental officials and industry representatives were interviewed.

The formal structure of the analysis borrows some elements from decision theory and decision analysis. (See [9,10]) Decision theory has developed a set of tools which can be used to aid decision makers in such complex tasks as setting and applying standards. Among such tools are structural tools (decision and goal trees), quantification methods (measuring intangibles, quantifying uncertainty), and evaluation methods (multiattribute utility methods).

For a descriptive analysis of the Norwegian and UK standard setting process the structural elements are probably as far as one can go. In the following sections the decision makers and other actors involved in the standard setting process will be described, including their goals, alternatives, and information linkages. Then the actual decision process will be discussed in terms of the information used to set standards and the institutional, engineering, and economic constraints and criteria. Finally, the paper will compare UK and Norwegian decision making and point out some problem areas.

2. ACTORS IN THE STANDARD-SETTING PROCESS

Standard-setting is a socio-economic and political process. This statement has been made previously by Majone [11] and Holden [12]. Majone has noted the following points in line with this thesis:

- a standard cannot be set on a purely scientific basis;
- a standard provides only an appearance of precision and hence of "scientific" character;
- a standard always represents an implicit evaluation of human well being;
- a standard is only one of other alternative means of regulation;
- the institutional framework often determines the decision on a standard;
- self-interest of regulatees moves them to attempt to modify the terms of the regulator, including any standards set.

In addition, Holden has noted that regulatory processes, including standard-setting, are based on a bargaining process between regulators and regulatees. [13] As an example of the political nature of standard-setting Schon notes that an attempt to set standards in the lumber industry for the "2x4" developed more political response than any other issue in the recent history of the US Department of Commerce. [14] Lumber producers both large and small, building interests, federal agencies, state governments as well as US Congressmen and Senators were all engaged in attempting to influence the thickness standard of the "2x4".

In regulatory decision making no simple decision making framework can be readily adopted, since a number of decision making units are involved. A complete system of actors that could be included in a regulatory process may represent:

- regulatory actors,
- developers,
- experts,
- impactees,
- exogenous actors.

For the present discussion only some aspects of these complex "decision making units" are emphasized. The exogenous actors, usually international institutions, often function as a constraint in regulatory decision making (e.g., the necessity to meet international standards). The experts function in their capacity to provide information pertinent to the regulation problem (e.g., information on biological effects of oil discharges).

Figure 1 shows an attempt to delineate the potential actors capable of being involved and impacted upon through chronic oil discharges from offshore platforms and tankers in the UK. For Norway (Figure 2) many of these actors are the same even though some of the governmental actors have slightly different names and responsibilities.

Before discussing the single actor groups in detail some interlinkages between actor groupings should be mentioned. As is readily discernible the development actors attempt to influence opinion and regulation in both the international (exogenous actors) and national (regulators) spheres. They have created the Exploration and Production Forum (E&P Forum) and Offshore Operator's Associations to perform the function of generating an informal network into related policy formation systems. In addition, other coordinating roles are occupied by the Central Unit on Environmental Pollution (CUEP, UK) and the State Pollution Control Agency (SPCA, Norway) which act as information sources to the oil-related Energy and Trade Ministries and as coordinating bodies in discharge control functions. The CUEP also acts as liaison for environmental matters at the international level.

One could expect a priori that each of the five actor subsystems would be tied together for the setting of standards for offshore chronic oil discharges. The following discussion will show that the actual actor configuration, their interlinkages and contacts are in fact more limited. In the discussion a distinction has to be made between the setting of guideline standards and the actual day-to-day process of reviewing applications to discharge oily water. In Norway, the standards are set almost exclusively on a case-by-case basis. In the UK, however, prior to individual platform arrangements an attempt was made to create some guideline standards that would provide the basis for day-to-day regulations.

2.1 Regulatory Actors

Both governments attempt to organize their environmental standard-setting process to interconnect with the offshore developers. Both countries have similar agencies to accomplish their tasks. The major difference is that the UK relies on its Petroleum Production Division to regulate offshore oil discharges while Norway uses its State Pollution Control Authority.

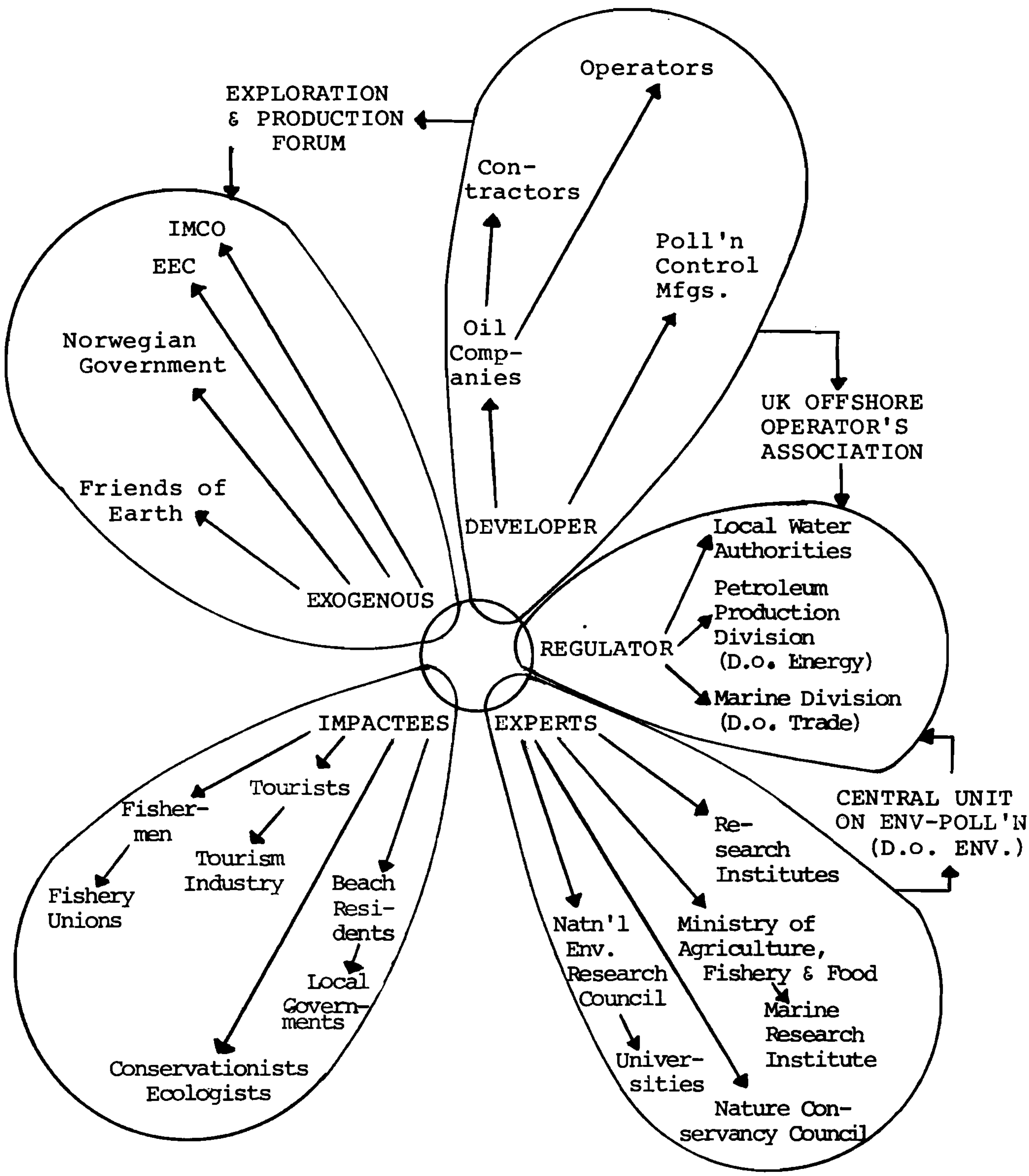

Figure 1 POTENTIAL ACTOR CONFIGURATION IN SETTING STANDARDS FOR OFFSHORE CHRONIC OIL DISCHARGES IN THE UK

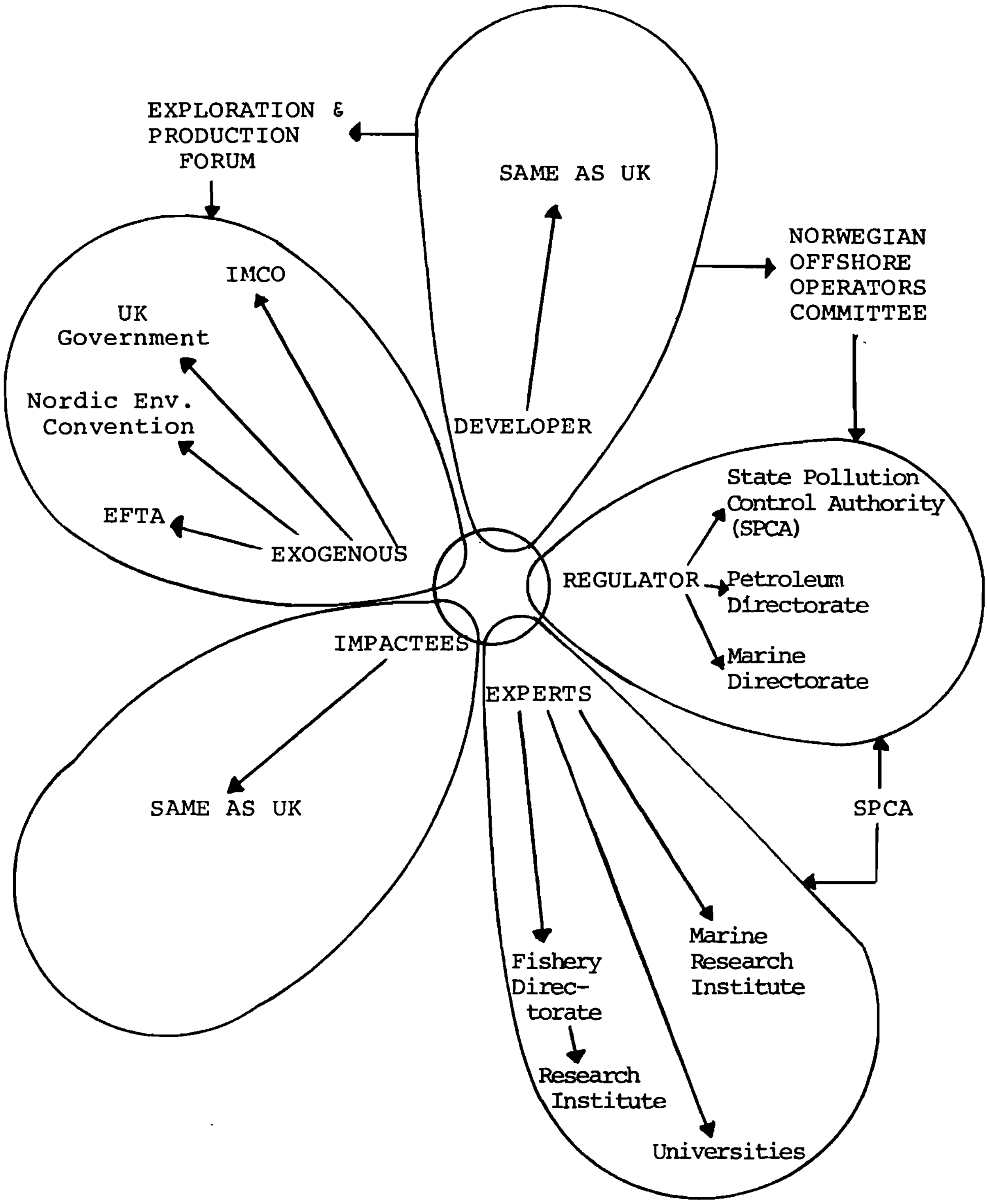

Figure 2 POTENTIAL ACTORS CONFIGURATION IN SETTING STANDARDS FOR OFFSHORE CHRONIC OIL DISCHARGES IN NORWAY

In the UK the CUEP performed the bulk of the work for studying the chronic oil discharge problem and for working out guideline standards. The main partner with which the CUEP consulted during the process was the Petroleum Production Division (PPD) of the Department of Energy, the division responsible for implementing offshore discharge standards. Contact with local regulators and authorities existed through the Scottish Office and Her Majesty's Industrial Pollution Inspectorate for Scotland. The Department of Trade was consulted in the standard-setting process through its Marine Division which is responsible for oil pollution from ships.

The Petroleum Production Division of the UK is in charge of the day-to-day implementation and adjustment of standards (the exemption to discharges under the 1975 act). In the process it interacts with a variety of government actors, including the CUEP, the Marine Division of the Department of Trade, as well as several expert actors such as the Ministry of Agriculture, Fisheries and Food and other divisions in the Department of Energy. However, this process takes place exclusively in-house in an interaction between key governmental actors.

In Norway the State Pollution Control Agency is responsible both for setting standards and for implementing them. As of now Norway has not decided to adopt firm standards for offshore platforms but deals with discharge applications on a case-by-case basis. Similar to the link between the CUEP and the PPD, the SPCA interacts closely with the Petroleum Directorate and to some extent with the Marine Directorate in its regulatory activity. However, unlike the PPD, the SPCA also provides information about discharges from proposed platforms to non-governmental experts, impactees, and exogenous actors on a routine basis.

The main difference in the regulatory actors involved in standard-setting between the UK and Norway is that in Norway the central actors is the SPCA, an environmental actor; while in the UK the main responsibility is in the hands of the PPD, an energy actor aided in its research by the CUEP, a unit in the Department of Environment. Furthermore, in the routine application of standards to platforms the SPCA has a wider range of linkages with actors outside of the industry-government sphere than the PPD. Thus the UK has placed the standard-setting process for offshore environmental protection directly into the agency that has the responsibility for overseeing continued petroleum production from offshore. Norway, on the other hand, has sought a counter-balance in its standard-setting process where an environmental agency regulates oil company pollution instead of an oil production agency.

2.2 Development Actors

The oil industry is the generic term for various development actors involved in the standard-setting for chronic oil discharges in Norway and the UK. The large private oil companies such as BP, Esso, Total, together with the national oil producers, the British National Oil Corporation and the Norwegian STATOIL, form the main body of what would fall under the heading of "developer". Included also are the various contractors (equipment producers, shippers, etc.), non-company operators and, in particular, the treatment equipment manufacturers.

In CUEP's research on determining the basis for offshore standards the oil industry was involved at several stages. First of all, they provided data to the CUEP about equipment availability, equipment performance, etc. Treatment manufacturers were contacted for this CUEP study, to provide data on costs and performance ranges of oily water treatment equipment. Second, they were involved through representatives in a public seminar which was held at Heriot-Watt University [15] in which the main issues related to chronic oil pollution standards were considered. Yet the oil industry feels that their involvement in setting these guideline standards was limited.

The oil companies and platform operators are in direct contact with the respective regulation agency (SPCA in Norway and PPD in the UK) in questions of oily water discharges from a specific production platform. The companies submit a design proposal or equipment specification with operating characteristics to the regulator for approval. This is then reviewed, and either accepted or modified. Furthermore, the SPCA and PPD are in contact with treatment manufacturers, when it comes to specific questions relating to treatment or monitoring equipment for a specific platform.

2.3 Expert Actors

Various governmental and non-governmental experts are involved in the standard-setting process both in Norway and the UK. In the work of the CUEP on chronic oil discharges the Sea Fisheries Laboratory of the Ministry of Agriculture, Fisheries and Food (MAFF) provided expertise about the possible effects of chronic oil pollution on fish and other marine organisms. The Warren Springs Laboratory provided technical expertise on oily water treatment. University researchers and expert councils, such as the Nature Conservancy Council or the National Environmental Research Council, were not directly involved in the analysis of chronic oil pollution by CUEP.

Experts are to some degree involved in the case-by-case standard-setting in the UK and Norway. The PPD sends the discharge application of the operator to various governmental groups, including MAFF, Warren Springs Laboratory, and the Marine Division of the Department of Trade. These applications are reviewed and commented on by experts from these Departments. The SPCA sends applications for discharges not only to governmental experts, but also to fishery union experts, and non-governmental environmental experts.

Thus the main difference between the UK and Norway with respect to the involvement of experts in chronic oil discharge standard setting is that the PPD relies solely on in-house expertise, while the SPCA also includes experts from outside of government. The confidentiality of the operator's report as well as the desire to maintain a good working relationship was mentioned as the main reasons for keeping the review within governmental agencies in the UK.

2.4 Impactee Actors

Potential sufferers of chronic oil pollution include the fishermen and possibly the consumers of fish. Also included are coastal residents, tourists, the tourism industry, and local governments who may suffer from oily or tarred beaches. A different type of impactee are ecologists, conservationists and environmental groups who may not be directly impacted upon, but their values (high evaluation of rare species, birds, etc.) are affected.

In the CUEP research on guideline standards these impactees had a possibility to be heard at a public seminar held by the CUEP, but they were not directly involved in the process of elaborating discharge standards. In the review process by the PPD the impactees are not involved, although some governmental experts could be interpreted as representing their personal or professional interests (e.g., biologists from the MAFF). In the Norwegian review process some impactees are involved; for example, fishery organizations receive discharge applications for comment.

2.5 Exogenous Actors

Exogenous actors involved in the basic work on standards by the CUEP were the SPCA and IMCO. Both were informed but not directly involved in the standard setting process. The EEC played a special role as an exogenous actor, since some member countries which share non-oil North Sea resources could potentially be affected by chronic oil pollution. The EEC has pushed for rather stringent across-the-board standards.

In terms of the routine review of discharge exemptions exogenous actors play a more limited role. The PPD and the SPCA have contact but do not inform each other about every discharge exemption case. There are contacts between the SPCA and the Nordic environmental convention, but it is not clear to what extent these contacts include actual discharge application reviews.

Although the above sections indicate all five actor groups were involved in the setting of standards and in the day-to-day review of discharge applications, the "core actors" are the regulator and the developer. Although the impactees have little direct involvement in the standard-setting process, the consideration of their interests by the regulator makes them an implicit part of the process. In the further analysis of the objectives, alternatives and decision making processes this paper will therefore concentrate on these three groups.

3. GOALS AND OBJECTIVES OF REGULATORS, DEVELOPERS AND IMPACTEES

The area of value and objectives has a much less firm data base than do the alternatives noted in the next section. It is usually difficult to elicit from the literature or from more or less informal discussions a good picture of goals and objectives for actual decision making. Therefore, some of the following arguments will be rather hypothetical.

3.1 Regulatory Objectives

Environmental control agencies of the UK and Norway list at various places goals and means for meeting environmental objectives. Table 3 shows that the UK and Norway have relatively similar goals and means for meeting environmental objectives. Both the UK and Norway give credence to the best practicable and best available means for meeting their environmental goals respectively. The UK defines "best practicable" as the ability to prevent or control pollutants as far as is practicable regarding local conditions, financial implications and current technical knowledge. [16] Best practicable is seen as distinct from the best technological since it accounts for economic and other implementation problems. Norway defines "best available" as the use of the best available technology known internationally but constrained by the costs of such technology. [17] Thus both countries virtually end up with the same definition. [18]

Further regulatory objectives are related to the pressures and constraints put on the regulator from the legal system and from international and administrative demands. Qualitatively speaking any regulation should:

- be easy to manage (implementation and control);
- fit into international agreements and rules;
- fit into the national legal and policy framework.

In the UK case such objectives worked rather as constraints. There had been much concern about fitting discharge regulations into international rules and policies. For platform regulations the international community put pressure on the regulator in the form of international conventions that suggested the early adoption of rather stringent uniform oil discharge standards for the North Sea. The constraints set by the legal framework in the UK were largely defined by the Pollution Acts of 1971 and 1975.

In addition, regulatory constraints were imposed through national oil development policy which tended to prohibit regulations which would seriously interfere with the pace and scale of oil development defined by political bodies. In the UK the goal

Table 3 COMPARISON OF POLLUTION CONTROL GOALS AND MEANS FOR THE UK AND NORWAY

UK POLLUTION CONTROL (1)		NORWAY POLLUTION CONTROL (2)	
GOALS	MEANS	GOALS	MEANS
• protection of human life and health	• use of best practicable means	• protect capacity of nature to produce and renew itself	• use of best available means
• safeguarding plant and animal life useful to man	• avoidance of rigid quantitative standards	• ensure pollution does not damage human health or affect personal well-being	• discharge may not occur without permission
• restrict interference with man's normal use of environment	• delegation of actual enforcement to local levels	• consider pollution control in relation to other social goals, including costs of such control	• pollution control included in cost of goods and services
• minimize permanent adverse effects on environment	• extensive monitoring for pollutants, humans and plants and animal species	• ensure similar goals are set in other countries	• pollution to be controlled at source using standards, land use controls and impact assessment

(1) Taken from: A.J. Fairclough, The UK Approach to Environmental Matters, speech given to B.P. Environmental Forum, London, December 1, 1976.

(2) Taken from: Ministry of Environment, Parliamentary Report No. 44 on Pollution Control Measures, Oslo, 1975-76.

See also: UNEP, Environmental Conservation in the Petroleum Industry, UNEP/ISS.5/10 Industry Sector Seminar, Paris, June 10, 1977, pp.41-43.

of rapid oil development could probably be considered an integral part of the regulatory objectives since the regulator is part of the Department of Energy.

In Norway the SPCA objectives may be considered more concentrated on the environmental side. Neither international agreements about pollution nor national energy policy seem to have had a large impact. However, the manageability criterion may have played a larger role in Norway which does not have a large research or regulatory capacity to set standards and enforce them regularly. The regulation of offshore platform discharge standards in Norway is still a one-man operation in the SPCA.

3.2 Developer's Objectives

Two objectives may be of over-riding importance to the developer: minimization of investment and operational costs for pollution control equipment and minimization of possible sanctions or costs due to violations of regulations. These objectives are in direct conflict. Good equipment is expensive but insures against violations of regulation or possible compensations to impactees.

While the cost of pollution control is easy to quantify the cost for violation of regulation are not clear cut. However, it was clear from discussions with environmental control officers of the oil companies that the oil industry was highly concerned about possible violations of governmental regulations. The effects of such violations are, of course, not only monetary in terms of additional costs for equipment, penalties, shut-down costs, etc., but they also relate to a worsening of the working relationship with the regulatory agencies.

The environmental control officers which were contacted in this study also pointed out that it was in the oil companies' interest to minimize oil pollution from production platforms. It is not clear, however, how this objective actually enters into the offshore operators' decision making about pollution control equipment and operation. It is probably fair to say that, although the oil companies' concern about pollution is genuine, it enters into the actual decision making rather through considerations of compliance with regulation and general public sentiments. The immediate environmental concern is therefore discussed in detail only for the impactees.

3.3 Impactees' Objectives

The fishery side is concerned about minimizing a possible loss of fish catch due to chronic oil pollution and about minimizing the risks of chronic toxicity and tainting of fish. An additional objective may be to minimize dirtying of equipment. Tourists, tourism industry, local governments and beach residents have identical objectives: preventing oil and tar balls from reaching the shore waters and amenity beaches. The public at large as consumers of fish and other marine organisms should have virtually identical interests as the fishing industry. Environmentalists and ecologists are impacted more in their perceptions and values and therefore have the additional objectives of minimizing ecological disturbances due to chronic oil pollution as well as minimizing mortality risks to rare marine species.

Figure 3 puts these objectives together under the explicit definition of the UK objectives set out by the Department of Environment. [19] The basic structure of the impactee objectives is similar in Norway, although the general values and sentiments of the "Norwegian way of life" may shift some of the priorities as compared to the UK case.

4. ALTERNATIVE REGULATIONS, TREATMENTS AND USES OF THE ENVIRONMENT

Each actor group has its specific set of alternatives: regulation alternatives, development alternatives, and environmental responses and alternatives (excluded here are the impactees whose perceptions are impacted by the development but who do not actively use the environment).

4.1 Regulatory Alternatives

Figure 4 presents in a logical decision tree some of the alternative means to regulate chronic oil discharges from offshore production platforms. It also shows the path that the UK and Norway have taken and where they ended. These alternatives span the widest range that could possibly be considered by the respective regulators, the PPD and the SPCA.

The actual alternatives considered were, of course, much more limited. In the research on chronic oil pollution control by the CUEP the main focus was on average and maximum emission levels, that is, on the required average performance of the oily water treatment equipment and the maximum which should not be exceeded a given percentage of the time. The question that the CUEP addressed in its research was mainly: which average and maximum level appears reasonable under environment, engineering, economic and political considerations? Extreme alternatives such as no regulation at all or of a total prohibition of discharges were never seriously considered.

The Norwegian SPCA considers standards on average performance in connection with total effluent volume as the main regulatory tool for oil pollution; however, in its case-by-case application of regulations to platforms it reverts, in practice, to an equipment specification and approval regulation. Maximum performance may be considered by the SPCA in the future.

Several other alternative regulatory means arise in such case-by-case regulation. These include:

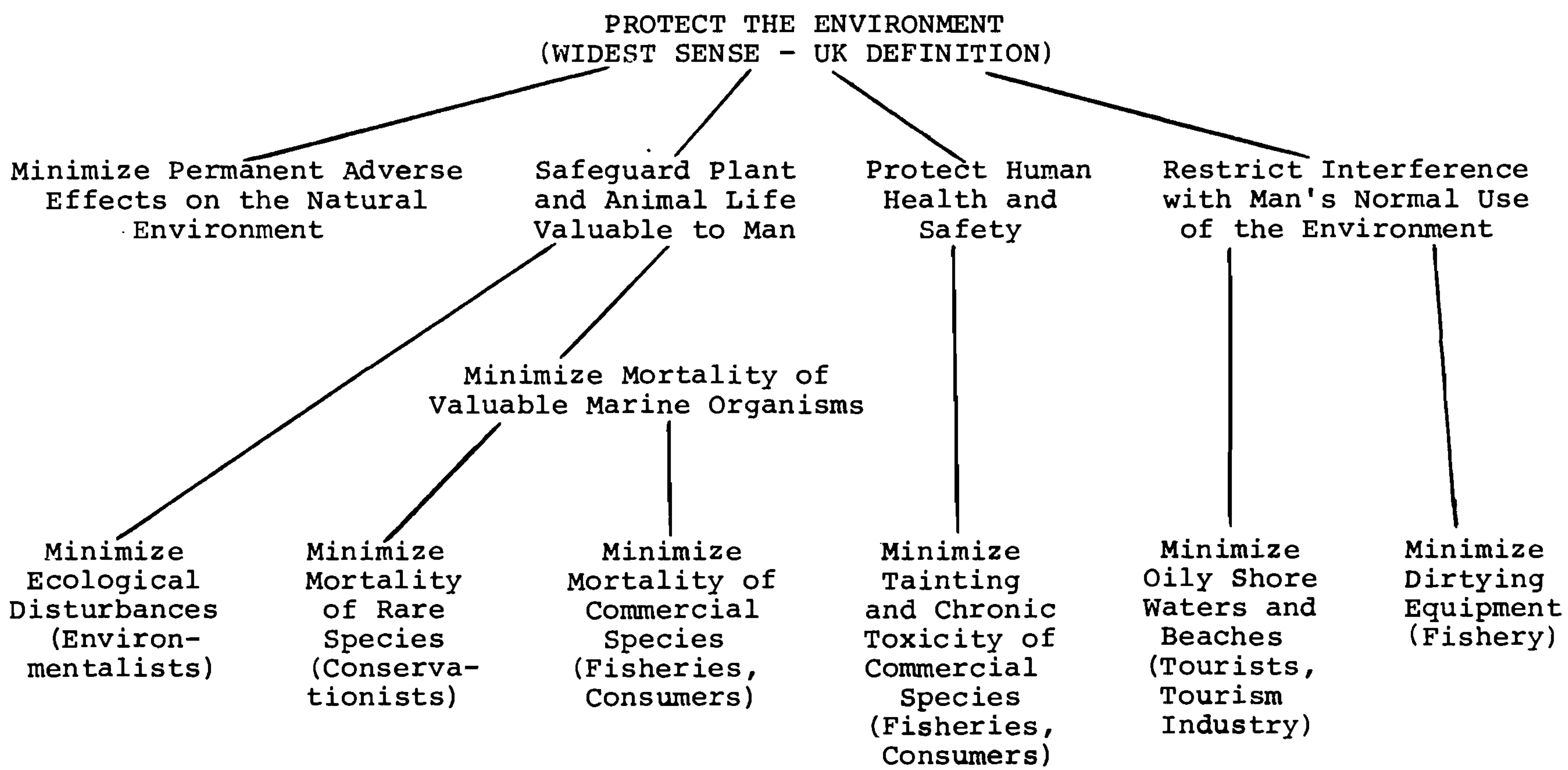

Figure 3 GOAL TREE FOR IMPACTEES CONCERNED ABOUT CHRONIC OIL POLLUTION

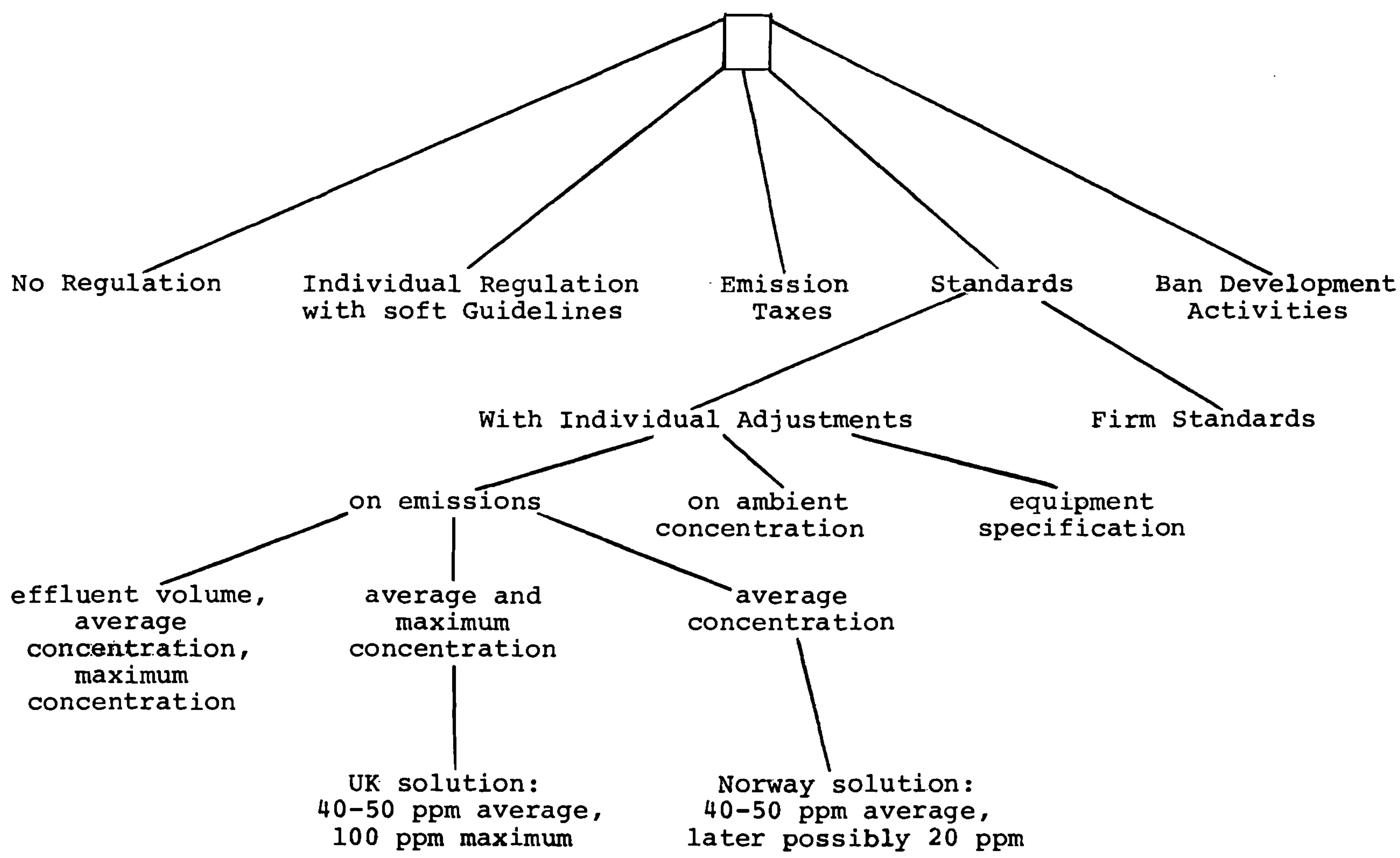

Figure 4 <u>TREE OF REGULATION ALTERNATIVES</u>

- the information requested by the oil company for the discharge application (e.g., cost, estimated performance figures, effluent volume, location of discharge, distribution of discharge, monitoring procedures, etc.);
- the circulation of applications for review (e.g., should non-governmental groups be involved or not);
- choices of the monitoring and inspection procedure to be imposed on the oil company;
- alternative sanctions for noncompliance.

4.2 Developer's Alternatives

Turning now to the substantive decision alternatives of the offshore operator, Figure 5 presents a logical decision tree including the kinds of treatment processes, equipment, and operating rules to run that equipment. In addition to the choices outlined in Figure 5, the operator has an additional choice not to discharge the oily water at the platform, but rather to pump it by pipeline to the shore where it can be treated or discharged. The primary treatment branch which is followed through in the figure is the most typical for offshore platforms. Sometimes it is coupled with secondary gas flotation or filtering equipment. For details of the equipment, the reader is referred to [20].

It is interesting to note that the oil companies virtually excluded the extreme alternatives from their considerations; that is, either no treatment at all or very severe treatment (biological or pumping oily water to the shore for treatment). Plate interceptors have become good operating practice all over the world and the offshore operators accept them as a practical way to treat oily water discharges.

Biological treatment is considered infeasible for offshore platforms, both by the operators and by the regulator. [21] The reason mentioned is the large size of biological treatment facilities (up to 4000 m^3 for a treatment volume of 10 000 tons per day). It is, however, not clear whether biological treatment is technically infeasible or whether it is feasible only at a very high cost. Concrete platforms with large storage volumes up to 100 000 tons should, in principle, be adaptable for biological treatment. Also it should be feasible from an engineering point of view to build a separate platform for a biological treatment plant, of course, at a substantial cost increase.

4.3 Impactees' Alternatives

The impactees alternative uses of the marine environment do not span a wide range. Fishermen can pursue either "business as usual" or divert fishing activities away from potentially polluted areas. The likely action is not to divert since it appears that fish are attracted by platforms. However, safety zones are established around platforms to avoid direct conflicts between these activities.

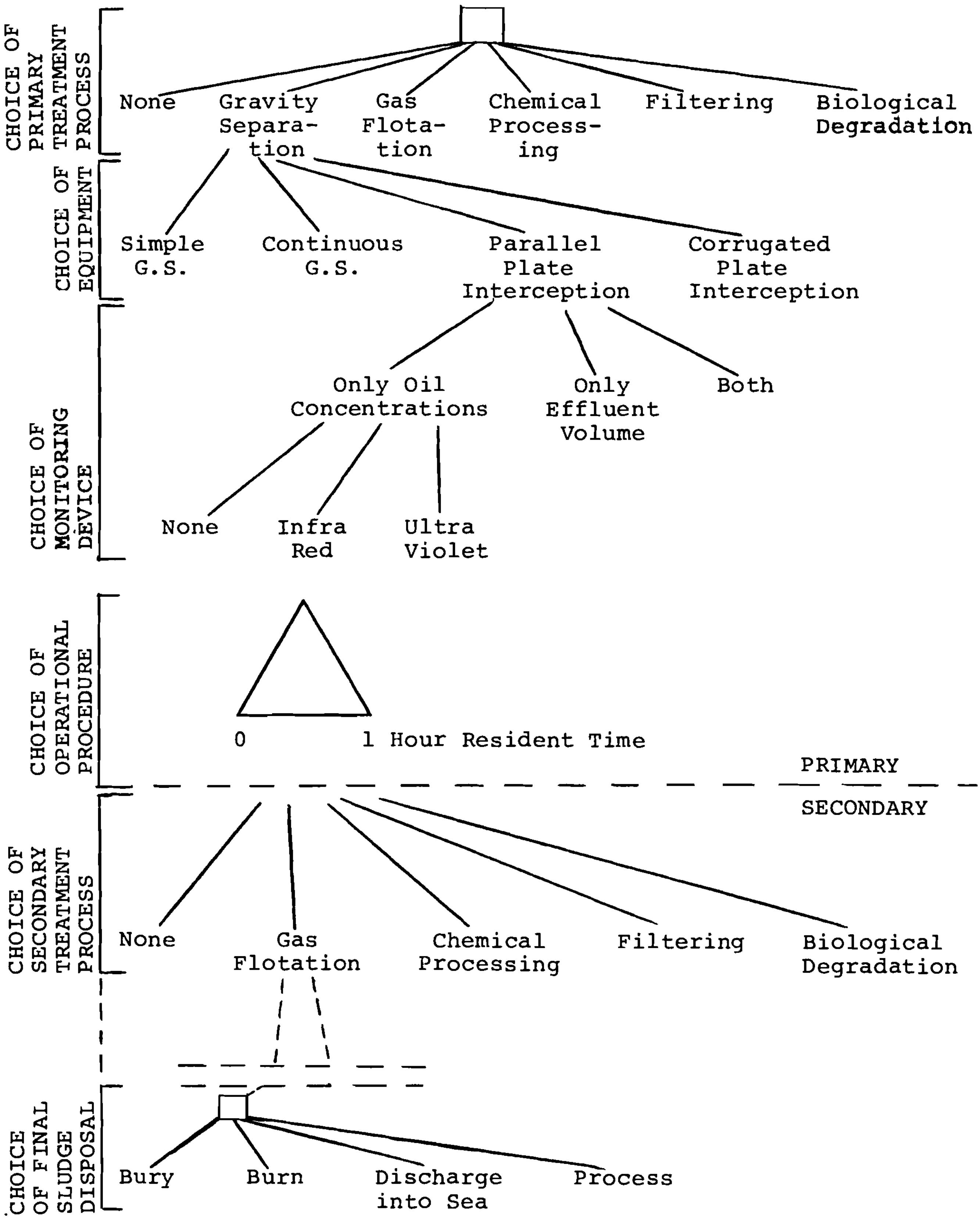

Figure 5 SOME BRANCHES OF A DEVELOPER'S TREE OF OILY WATER TREATMENT ALTERNATIVES

From present data and the scale of development it does not appear that shorelines are affected by chronic oil pollution from offshore platforms. Therefore, it is unlikely that tourists or beach residents would have to take recourse to such extreme alternatives as to move to non-polluted areas.

Although all groups have, in principle, the additional alternative of putting pressure on the regulator and the developer either through direct petition or through various forms of public and legal action, no such actions seem to have been considered yet by any of the impactees.

5. THE DECISION PROCESS IN STANDARD SETTING

While the previous sections described and discussed the elements of standard-setting for chronic oil discharges in terms of actors involved and their objectives and alternatives, the following section will describe the linkages in terms of information flows, procedures, and data base in the actual decision process for setting chronic oil discharge standards.

5.1 UK Standard Setting

Although the PPD issued discharge exemptions on a temporary basis as soon as the first production platforms began operations, the actual standard-setting process began only in 1975 with the research done by the CUEP. This research aimed at identifying guideline standards which would satisfy the environmental policy criterion of enforcing "best practicable" means of pollution control, in line with engineering, economic, environmental, and political objectives.

It soon became clear to the researchers involved that the uncertainty about biological effects made the setting of discharge standards on a purely biological basis virtually impossible. The fate, turnover, and effect of oily water emissions have been reviewed by several authors [22,23,24] but no firm conclusions can be drawn. In the North Sea it appears that effluents with a concentration of 50 ppm or less generate ambient oil concentration levels which are hard to detect against the background hydrocarbons in the sea water. Even at a highly polluting platform in the Ekofisk field, no noticeable oil concentrations were found away from the emission source.

Some marine biologists would agree that reasonable effluent volumes with 50 ppm oil concentration and with good dispersion characteristics will not lead to any direct harmful effects on marine organisms. Some oil company experts argue even stronger. From their experience in the Gulf of Mexico no short or long term effects could be shown even at higher effluent concentrations. Independent marine biologists are more cautious: they leave open the possibility of long term cumulative effects of low oil concentration in water or short term and medium term behavioural effects such as on spawning behaviour.

Because of this uncertainty about biological effects of oil in the sea water, the CUEP concentrated its effort mainly on estimates of total amounts of oil discharged in the North Sea and on technical feasibility, costs, and performance of oily water treatment equipment. By making rather stringent assumptions about treatment levels (25 ppm), production rates, transportation, and storage the CUEP calculated that approximately 585 tons of oil will be discharged from UK platforms in 1981. [25] In this study an independent estimate of 2,200 tons per year was made for the whole North Sea from all production platforms. The difference cannot be explained by the additional platforms in the Ekofisk and the Statfjord field alone. It appears that the UK estimate is based on far too optimistic assumptions. Even the estimate of this paper may be below actual performance figures. Platforms exist in the Norwegian sector which are estimated to discharge more than 800 tons of oil per year at peak production levels.

On the technical and economic side the CUEP studied in detail various oily water treatment equipment. A summarized version of the results is presented in Table 4. The performance figures could only be given in ranges since equipment manufacturers, oil companies, and independent researchers differ, often substantially, in their assessment of equipment performance. This uncertainty about actual performance played, in fact, a large role in standard-setting. Equipment manufacturers typically cite very good performance figures, while the oil companies doubt that these figures can be achieved in the field. In discussions with oil company representatives views were expressed that the figures used by the CUEP may be too optimistic. They feared that standards set on the basis of such optimistic figures could not be achieved in practice.

The study by the CUEP naturally did not mention any political objectives in setting standards. However, it should be clear that international pressures would have required standards set in the range of international acceptability (currently such standards vary from 10 ppm in Japan to 50 ppm in the US).

The CUEP study implies that standards of 30-40 ppm should be achievable with currently used technology at a reasonable cost. The conclusion of the CUEP study was, among others:

> "Data suggest that an average oil concentration of 30-40 ppm is achievable with present technology. A maximum effluent oil concentration of 100 ppm should not be exceeded more than 2% of the time. New developments of existing systems for use on platforms may be able to reduce the effluent oil concentration to an average of 20 ppm, but these systems have not yet been fully evaluated." [25, p.22]

The translation of these guideline standards into the case-by-case standard setting for individual platforms is done by the PPD. An application for an exemption to discharge is forwarded to the PPD from the oil company wanting to build the platform. Information is requested by the PPD from the company such as

Table 4 ASSESSMENT OF VALUE RELEVANT CONSEQUENCES FOR THE MAIN TREATMENT ALTERNATIVES

(from Pollution Paper No. 6, Department of Environment, UK)

TREATMENT ALTERNATIVE	COST RANGE (INSTALLMENT FOR 10 000 ton unit)	PERFORMANCE RANGE (ppm)		FEASIBLE ON OR OFFSHORE
		AVERAGE	98% OF TIME	
Simple Gravity Tank	?	40-50	155	BOTH
Corrugated Plate Interceptor	US$ 130,000-175,000	10-50	?	BOTH
Gas Flotation	US$ 175,000	27-34	100	BOTH
Filters	?	21-22	60	ON
Biological Treatment	?	1	?	ON ONLY

monitoring data from around the proposed platform site, the production aspects of the platform, volumes of oil to be treated, and location and depth of discharges. This information is then summarized and circulated to the organizations as described earlier, all of which are within the UK government. Comments received are then used as the basis for the PPD in setting oil discharge standards.

Contacts between the oil company and the regulator are informal. If the application is not acceptable as is the regulator can only suggest problem areas and cannot tell the company what it should do to correct the situation. This problem exists because the regulator may have to advise the Secretary should the company decide to appeal the regulator's ruling. Thus the PPD only recommends the limit of discharge (maximum average, maximum to be exceeded four percent of the time, maximum volume per day, etc.,) and the conditions necessary to obtain an exemption. Monitoring procedures are determined by the PPD. The granting of the exemption and the conditions under which it is granted are also circulated to the same organizations for comment. Again standards are set on a case-by-case basis. Information on such standards and discharges is available to the public ex post facto in the annual reports published by the Department of Energy.

The PPD in conjunction with the CUEP has evolved six basic operating principles:

1) provision of certain basic information on oil treatment alternatives on offshore platforms to oil companies,

2) emphasis on pollution control equipment since the effects of oil in the marine environment are not well known,

3) emphasis on good pollution control equipment being installed to meet the agreed standard of discharge,

4) maintenance of good informal working relations among government departments and between government and companies,

5) setting of good monitoring procedure as part of the conditions to obtain an exemption for oil discharges,

6) burden of proof for not installing treatment devices is shifted to companies since they must apply for exemption.

In January 1977 eight production platforms operated in the UK sector, six of which produce oily water discharges. The exemptions which have been granted so far were generally based on a standard of 40 ppm average oil concentration for production platforms with a large discharge volume (greater than 100 000 barrels of oily water a day) and a maximum of 100 ppm not to be exceeded 96% of the time. Platforms with smaller effluent volumes are required to treat oily water down to a level of 50 ppm average oil concentration. One platform has been granted an exemption although its oil concentration is on the average higher than 50 ppm.

These decisions indicate only slight deviations from the CUEP proposal. Most notable is the change from 98% to 96% for excess maximal performance. It is unclear by what process this change occurred.

Monitoring is used in the UK to provide a data base for standard-setting. Oil companies are given responsibility for such monitoring. Each platform is to be monitored twice a day by the company having operational responsibility. The PPD determines the method of monitoring and the method of analysis. Twice a month the platforms are inspected by the PPD, but since helicopter space must be reserved in advance no surprise inspections are possible. The results of such monitoring are subjected to analysis by a government laboratory. The expenses of such analyses are borne by the companies involved.

5.2 Norwegian Standard Setting

In the Norwegian case no research was carried out to determine guideline standards. However, several reports to the Storting, the Norwegian parliament, [26, 27] indicate the positions which the Norwegian government takes with respect to oil discharges and possible environmental effects. As mentioned before, the estimates for total oil discharges from North Sea oil development are more pessimistic than the ones generated by UK governmental researchers. Also the overall impression from these reports is that the potential environmental effects of oil are taken much more seriously than in the UK. In fact, environmental considerations had a strong influence on the Norwegian decision to halt development North of the 62nd parallel.

The SPCA sets and revises standards for offshore platforms against this background on a case-by-case basis, revising its position as the process goes along. The SPCA requests information from the oil company wanting to discharge oily water from offshore drilling and production platforms. The oil pollution control equipment used to reduce oil discharges is included as part of the application for the platform. The information requirements include:

- production process: amounts of oil to be produced, how the oil is to be handled, where oil pollution points exist, amounts of other wastes to be produced;
- monitoring process: how marine environment around platform is to be monitored;
- location of platform offshore (location is non-negotiable);
- function of platform: drilling for what kinds of petroleum, how petroleum is transported;
- storage of oil: where and how;
- effluent treatment: displacement water, production water, drainage water collection and treatment plus treatment costs.

The above information is then summarized, shown to the oil company first, and then sent to the organizations noted earlier for comment within eight weeks. Once comments are received they are used to guide the SPCA in its standard-setting process. Contacts between the oil company and the SPCA are informal, and they meet together to discuss the tentative standards before they are finally set. When the SPCA meets with the oil company both the Oil Directorate and the Marine Research Institute are invited.

Generally, discharges are allowed if the operator uses equipment that reduces the oil content in the effluent down to 50 ppm. However, the SPCA considers tightening up this standard in the future. Although no firm data are available about actual performance yet, it appears that the total volume of oil discharged may be substantially higher in Norway than in the UK due to large volumes of displacement water discharges. Also, some platforms in the Ekofisk field still use relatively poor treatment equipment.

The SPCA has adopted five basic operating assumptions:

1) a wide circulation of information of pending platform applications even though little hope exists that the impactee or exogenous actors will have information of use to the regulator,

2) an emphasis on pollution control equipment rather than on standards per se since the effects of oil in the marine environment are not well known,

3) an emphasis on working with the oil companies in the platform design stage to allow pollution control equipment to be integrated into the platform design itself,

4) if pollution can be technically controlled the SPCA is interested in reducing such pollution if at all possible and reasonable to do so,

5) given the lack of information on effects of oil in the marine environment the environment is given a strong bias to preserve future options.

One particular problem for the SPCA is its relative newness in regulating offshore oil activities. It has had little experience in dealing with multi-national companies. It also does not have the manpower and information resources available to it that such companies have. Therefore, its knowledge of alternatives is reduced and it attempts to depend on other bodies for information and for valuations of such information.

In addition, monitoring procedures are established. The oil companies are asked to do baseline monitoring in the vicinity of the platform location before construction occurs in cooperation with the Marine Research Institute. During the actual production phase monitoring continues at fixed points around the platform, again the company taking primary responsibility in cooperation with the Marine Research Institute. Only levels of oil discharges are collected. These figures and samples are then sent to be evaluated by the Marine Research Institute and the Fishery Research Institute. The former institute also conducts an independent monitoring program on oil levels monthly. The complexity of hydrocarbon compounds and the relationship of oil with other pollutants makes evaluation of monitoring results (as well as setting standards) difficult. Also companies are reluctant to do continuous monitoring and then send such data to independent institutes for fear that their own figures would be used against them. The greatest problem in Norwegian monitoring appears to be a lack of a common monitoring framework using a wide variety of biological and ecological variables tied together into a systems approach.

5.3 Biological Information for Standard-Setting

While both countries have adopted monitoring programs as a part of their standard-setting process it is clear that a major gap exists in the availability of adequate information on the long-term effects of chronic oil discharges on marine organisms. One would like to answer questions for such a consequence assessment as: Given normal fishing operations and no oil development what is the amount and quality of fish catch that the fishing industry can expect? To what degree do different levels and amounts of chronic oil pollution reduce the amount and quality of the catch? To what degree does chronic oil pollution endanger the ecological balance in the marine environment? To what degree does chronic oil pollution contribute to dirty fishing equipment and to producing dirty shore waters and oil slicks on amenity beaches?

The basic answer to these questions from scientists, regulators and managers was: we do not know. There seems to be an enormous uncertainty with respect to biological effects of oil that is discharged at low but constant levels into the sea waters. Ambient oil concentrations are hard to detect against background hydrocarbons in the sea. Even at a highly polluting platform in the Ekofisk field (at least 10-30 tons of oil are being discharged there monthly) which is located near a very large herring spawning ground, marine biologists can not claim that there are effects of oil on fish.

The process from emission to the fish or marine organisms into the food chain is uncertain: the fate and turnover of oil once released is currently beyond measurement. But in the judgement of marine experts reasonable effluent volumes with 50 ppm discharged offshore with good dispersion will not lead to any direct harmful effects on fish and marine organisms. On the other hand, marine biologists do not reject the possibility of behavioural effects of small amounts of oil pollution (e.g., on spawning behaviour) and of long-term cumulative effects on fish and in the food chain. An additional source of uncertainty arises onshore, where synergetic effects of oil together with other pollutants from refineries and other industries may occur.

The high uncertainty about the effects of biological effects created another source of opinion conflict among the various decision making units involved in the regulation problem. Experts from the offshore operators side claim no effects whatsoever with present treatment levels. Some marine biologists and fishery representatives warn of the potential long-term effects.

In the light of such uncertainty and the importance of this issue one would expect both countries to have staged a comprehensive marine research program on effects of oil pollution. Although the problem of oil pollution in the North Seas was known several years ago, up to now the research responses to reduce this uncertainty have been slow and rather ad hoc.

The main result of the uncertainty about biological effects in the Norwegian regulation setting was a rather pessimistic view. The pessimism with respect to consequence estimates has, of course, several other supporting sources: the strong Norwegian sentiment favouring fishing and the general policy of slowed development of the oil fields. The first sources makes a pessimistic assessment of biological effects a political necessity, the second factor allows the pessimism to be translated into rather strict standards and regulations. Practically, this pessimism is related to the environmental policy to use best technical (applicable) means to prevent chronic oil pollution.

In the UK regulation the uncertainty about biological effects resulted in a shift from environmental considerations in regulations to equipment performance and cost considerations. Rather than starting - like the Norwegian regulators claim - with a worse environmental case attitude, UK regulators asked which efficient treatment equipment can be implemented without imposing too high a cost to the developer. This consequence of the uncertainty about biological effects is compatible with the prevailing UK objectives of rapid oil field development and the general environmental policy to use "best practical means" in reducing pollution.

6. COMPARISONS AND CONCLUSIONS

Comparing Norway with the UK it is clear that certain similarities and differences exist in their standard-setting processes for offshore oil discharges from platforms. Table 5 is an attempt to summarize the differences between Norway and the UK. Since it is generally self-explanatory little necessity exists for elaborating on the information in this table. The similarities between these two countries include the following points:

- emphasize treatment equipment;
- standards tied to equipment;
- standards (equipment) set on a case-by-case basis;
- standards set in vicinity of 40-50 ppm;
- attempt to work with company in design stage;

Table 5 COMPARISON OF REGULATORS OF OIL DISCHARGES CONNECTED WITH OFFSHORE PLATFORMS IN THE NORTH SEA

DIFFERENCES BETWEEN NORWAY AND UK	
NORWAY	UK
• regulator is pollution control authority with emphasis on environment	• regulator is petroleum production unit with emphasis on energy
• regulator had to create regulatory structure	• regulator had to redesign regulatory structure
• impactees asked for comments	• impactees not asked for comments
• regulator open with information	• regulator keeps information confidential
• problem viewed as environmental with emphasis on fish impacts	• problem viewed as political and technical
• no research on treatment alternatives	• published one paper on treatment alternatives
• pessimstic estimates of total amount of oil discharges	• optimistic estimates of total amount of oil discharges

- monitoring process tied to discharge permit;
- responsibility on company to monitor;
- energy unit inspects monitoring.

One key finding is that both countries must wholly rely on entrepreneurial endeavour for oil discharge treatment alternatives. Neither country is attempting to initiate a research program to develop technological treatment alternatives; nor are these countries testing such equipment for performance ranges.

In addition, the standards set depend directly on such treatment equipment as is shown below:

- availability: existence
- capability: performance
- practicability: size and price.

Given a lack of information on biological and chemical effects of oil on the marine environment both regulators have turned to practical solutions to their regulatory function. Key factors include the platform space available for containing such treatment equipment on the platform as well as the cost of such equipment, which has been reported as approaching ten percent of total platform cost. Given this high cost and the lack of data on negative effects of oil in the marine environment it is surprising that the oil companies appear so willing to install treatment equipment. It would be very difficult for the regulators to state their case on a strong research basis.

Another important finding is that the degree of severity of oil discharge standards offshore affects oil discharges nearshore or onshore as well. If the offshore standard is low, say 0-20 ppm, and if it is coupled with a rigorous offshore monitoring, inspection and sanction program then one could expect that discharges might occur elsewhere in the company's operations closer into or onshore. Therefore, given a cost constraint to the company a tightening of operations in one place can mean a loosening of operations in another place which may have greater environmental effects.

The last major finding is the lack of a role for the actual quality of the environment in the setting of discharge standards offshore. In no case discussed did the environmental quality appear to be a significant factor in the standard-setting process. There appeared to be no fundamental change in location, design, technology or operation of any offshore project based solely on environmental quality criteria. Rather in each case standards were tied directly to performance of treatment equipment available for installation on platforms. Data other than for treatment equipment comes from physical aspects of the North Sea such as winds, currents, etc., rather than from any biological or ecological data.

Each of these regulators have quite different perceptions of the other. For example, the SPCA sees the UK as linking its standards to the assimilative capacity of the site whereby treatment equipment is used only when the expected pollution levels will exceed the capacity of that site. On the other hand, the CUEP and PPD see Norway as being very unreasonable in forcing companies to adopt the latest equipment available regardless of cost. As seen from the earlier discussion neither perception is correct.

The analysis of the standard-setting process about chronic oil discharges in the UK and Norway leads to several recommendations as to how the regulators could possibly improve their standard-setting process. While actual performance cannot be evaluated yet, the regulators' decision making intentions and capabilities have been identified and discussed.

In reviewing and updating oil discharge standards it will be necessary to continually monitor ambient distributions and effects on marine organisms. Therefore a comprehensive research program should be organized to assess changes in ambient oil concentrations close and far away from platforms and terminals. In addition, a stronger effort should be made to study the biological effects of oil in the North Sea. As a starting point, marine biologists should be encouraged to state and quantify what they know and what they do not know about biological effects and assess at least the extreme upper and lower limits of detrimental effects from offshore marine pollution. The data from a comprehensive marine research program could then be used to update the information already expressed in present knowledge.

Secondly, it would be advisable to maintain an ongoing file of possible regulatory and treatment options as reference points. At the moment the regulators are forced to evaluate only the technical design proposal made by the developer. The regulator himself does not have sufficient in-house research capacity to come up with specific alternatives to change proposed treatment plans. Consequently, most regulatory decisons will remain patchwork. If the regulator cannot maintain a file of possible treatment alternatives for platforms, the developer should be requested to submit several proposals at different discharge levels for the regulator to select among them.

Thirdly, the regulator should make more explicit his own objectives in dealing with oil discharges, both in terms of internal regulation objectives and in terms of trade-offs between development and environment. Measures of these objectives could then continually be used to analyse present standards and regulations. They could also be used to communicate between central and local regulators and to solicit more useful responses from reviewers in the discharge application review process.

Fourth, there may be some benefit in quantifying some of the intangibles, at least by marking them against reference events or reference alternatives. For example, one may think of qualitatively comparing the assumed biological effects of several amounts of oil spills with discharges from platforms, both against a "no development" alternative. The step from such a qualitative comparison to a judgemental scaling of effects is not very large. Such quantification could then be updated continually on the basis of new information and lead to faster responses in changing standards and regulations.

Fifth, public participation and advocacy in the standard setting and regulation process could be improved. By this is meant both a higher involvement (for example in the UK review process of discharge applications) and a more structured dialogue, (in eliciting preferences and opinions from environmentalists and fishery representatives). Public participation should be made part of the decision making process rather than operating mainly as an insurance against future complaints and criticism.

All of these suggestions amount to a more comprehensive and integrated system for environmental information collection, evaluation and decision making. While these proposals are costly, considering the highly uncertain future effects of chronic oil discharges, they may turn out to be less costly than future efforts and strains to avert a possible disaster.

The systems perspective of actors in a socio-economic-political process emphasizes comprehensive and integrated responses to environmental management. It is clear that such a response pattern would be quite easy to set aside since the results of a systems approach would not show in the immediate future, except for accident responses which are often soon forgotten by the public. However, the evidence in this study shows that the record is mixed. While core actors do dominate the decision-making system and do attempt to influence other parts of the system to their viewpoint it is clear that other actors are becoming more vocal. The actual persuasiveness of other actors is still open to consideration. Nevertheless, as core actors both the oil industry and the requisite government departments have recognized the necessity of expanding their information network, pressure points and feedback patterns.

The environmental management system in such a vast and unknown area as the North Sea is characterized by situations which call for a joint energy-environment systems response. The roles generated by the core actors of government and industry have worked well to implement the development goals of these actors, namely the maintenance of a steady development pace and increasing scale of development. However, this role has worked only partially well in the overall environmental area, despite efforts by interested parties in industry and government. The fact that impactees and outside experts have a low profile in decision making is shown through the mismatch of roles. Such issues as the above suggest that the decision making system itself may not only be lacking in such respects as including impactees from offshore oil development and non-government affiliated experts but that it is not responding comprehensively nor in any integrated way. A comprehensive technology assessment system would provide for the inclusion of all actor groups affected by offshore petroleum development. [28] Only those interests will be considered in any assessment when all interested or affected parties are directly involved in the assessment process.

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STANDARDS AGAINST NOISE POLLUTION: THE CASE OF SHINKANSEN TRAINS IN JAPAN

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STANDARDS AGAINST NOISE POLLUTION: THE CASE OF SHINKANSEN TRAINS IN JAPAN

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OUTLINE

ABSTRACT

This paper analyses the problem of noise pollution control for the super-rapid Shinkansen trains in Japan. The approach of the paper is a structured policy analysis based on an identification of actors, their objectives, and decision alternatives. Such actors include, the Environmental Agency of Japan, the Japanese National Railway Corporation (JNR), and the various residents groups fighting Shinkansen noise pollution. The structural part of the analysis elaborates a set of possible objectives and alternatives for pollution control, and discusses the decision making intentions of the actors involved. Two specific decision processes are then described in detail: the decision process of the Environment Agency to set noise standards for Shinkansen trains and the actions of the Association against Nagoya Shinkansen Public Nuisance against JNR.

Findings are derived from comparing the results of the structural analysis (possible objectives and alternatives, main decision making intentions) with the actual decision making. A main finding is that the Environmental Agency intended to set environmental quality standards (targets) but instead set enforcement standards (rules). These enforcement standards were evaluated and selected as if they were quality standards, that is purely on the basis of environmental information. Other information on the cost, engineering, national economic impact of regulation, which should have been considered for enforcement standards, was neglected. Another finding is the reactive role of JNR in the standard setting process as well as in the interaction with the Nagoya Association. JNR perceives Shinkansen noise largely as an engineering problem with engineering solutions responding to regulations. JNR's reactions were based on a desire to maintain the status quo (maintain train speed, comfort, safety) and to keep future development options open. The Nagoya Association had an impact on the decision making process of the Environment Agency and JNR only indirectly, since Japanese environmental decision making does not provide for direct inputs from impactees. The Association managed to enlist a variety of governmental and union supporters for their cases to stop Shinkansen noise, but failing to effectively interact with JNR and the Environment Agency, it finally began a court case against JNR which is still pending.

Overall, the findings of this study are a lack of comprehensiveness in the objectives and alternatives considered by the actors involved in the decision making and limited interactions between the main decision making groups. The discussion argues for using formal tools to improve standard setting decisions.

INTRODUCTION TO THE PROBLEM OF NOISE POLLUTION CONTROL FOR SHINKANSEN TRAINS

Like other types of pollution, noise can disturb, irritate, hurt and even damage. But few other pollutants cause such immediate and obvious aggravation as noise. The effects of noise range from slight disturbance of sleep or conversation, over physiological distress, to psychosomatic stress symptoms, and even physiological damage to the hearing system. There is another characteristic feature of noise pollution: people who suffer from noise usually quickly identify its source, complain, and try to stop it. In fact, in the last few years noise was the single largest source of complaints about environmental pollution in Japan. Statistics show that in 1974 noise clearly dominated air, water, and other types of pollution with 24,000 complaints. As one high official of the Environment Agency summarized: "Noise pollution, particularly of transportation pollution, has become the most important difficult problem at present" (Hashimoto, 1975, p.1).

Hashimoto especially singled out transportation noise because there has been massive action by citizens and political groups against noise from highways, railways, and airports. The most dramatic case was probably the battle against the new Narita International airport in Tokyo. Citizens and radical political groups have prevented the start of operations there for several years. Less dramatic, but with equally severe effects was a legal battle involving Osaka International Airport which citizens won in 1974. The court put severe restrictions on frequency and timing of jet flights. Residents also have begun protest action to fight highway noise. A law suit was filed in 1974 by the residents along the Kobe-Osaka Freeway. In this list of protest targets railways are not missing. In fact, the Shinkansen has been one of the most criticized sources of noise pollution in Japan.

Nobody denies that the Shinkansen is noisy. With its speed up to 210 km/hour it can reach noise levels around 110 dB*. Jet planes (from a good distance) and sledgehammers (from nearby) are louder, motorcycles are comparatively quiet. To give some reference points: the interior noise of some cars at high speeds can be around 80 dB, at busy street intersections the noise level is about 70 dB, in quiet residential areas at night the noise level may be 30 dB. Between 30 and 60 dB sounds are usually not

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* The decibel scale (dB) is the fundamental measure of noise which reflects approximately the human perception of loudness. Sometimes a weighted scale dB(A) is used which also takes different frequencies of noise into account. Other measures are the phone and the sone scale as well as physical measures of sound intensity. For details, the reader is referred to Beranek, 1971.

considered "noisy", unless the environment or the sound itself has some peculiar characteristic.

Residents in a distance of up to 100 meters from the tracks may be substantially disturbed by Shinkansen noise. Complaints about Shinkansen noise over the last few years confirm this. Table 1 shows the distribution of complaints and petitions against the Shinkansen between 1964 and 1975 on the highly populated stretch between Tokyo and Osaka. Noise and vibration clearly dominate the picture. But people and residents have not stopped with complaints and petitions. New lines of the Shinkansen like the Tohoku-Juetsu Shinkansen are sites of residents' protests. Residents have formed organizations such as the "League Against Shinkansen Pollution", and "Association Against Nagoya Shinkansen Public Nuisance", and the "Protest Group Against Shinkansen TV Interference", to represent their interests against Japanese National Railways (JNR). In one word: The Shinkansen has become a major target of environmental criticism and protest in Japan.

The growing conflict between JNR and residents caused regulatory agencies to act. Before the establishment of the Environment Agency in 1971, several regulatory and legal actions had been taken against noise pollution in Japan, primarily on the industrial and workbench noise sector. With the birth of the Environment Agency (EA) noise pollution control measures were enacted rapidly: in 1972, the Director of the EA recommended to the Ministry of Transportation and JNR to take "urgent steps on Shinkansen noise abatement".

Table 1: Complaints against Shinkansen Noise (Tokyo-Shin-Osaka)

Year / On	1964	65	66	67	68	69	70	71	72	73	74	75	75	Total
Noise	9	13	16	6	2	2	1	2	3	6	5	12	13	90
Vibration	11	16	19	9	3	5	1	2	5	1	6	9	0	87
Noise & Vibration	10	13	7	10	5	3	3	17	8	16	36	41	6	175
Wind Pressure	2	4	1	1									2	10
Interference on TV Others	1	4	4	6	6	1	6	6	1	1	11	10	10	67
	33	50	47	32	16	11	11	27	17	24	58	72	31	429

Source: Yorino, 1977b

At the same time a special committee in the Central Council for Control of Environmental Pollution (the advisory body to the EA) began to study the problem of noise pollution from transportation sources. Environmental quality standards of 70-75 dB were finally issued by the EA for Shinkansen noise in 1975.

But the noise standards and JNR's efforts to comply within the given time framework did not end the conflict between JNR and the residents. The "Association against Nagoya Shinkansen Public Nuisance" filed a law suit against JNR which is still pending; the plaintiffs did not accept the EA standards and still request stricter standards, slower speeds, and compensation. New construction sites of Shinkansens are again the issue of strong public debate. At present nobody can predict the outcome of this protest development. It appears that the noise problem, already a major problem for JNR as well as residents and the governmental regulators, may persist and become a main obstacle for further development of the Shinkansen.

In the following sections the decision process of JNR, the EA, and residents groups will be analysed to structure the decision problem of noise pollution control, to describe the decision process of the three actor groups, and to identify sources of conflict. The data for this study were collected during a three week field study in Japan. All major actors were interviewed in depth. No actual measurements were taken, and the quantifications reported are largely from the literature. An analytical approach will be used to structure the noise control problem which has already been applied to studying the problem on chronic oil pollution control in the North Sea (Fischer and v. Winterfeldt, 1978). The approach is a structured policy analysis of the decision process which generates a qualitative problem oriented picture of the pollution control problem by answering such questions as: Who is involved? What are the objectives? What are their decision alternatives? These structural elements of the decision making are then put together in an actual description of the information processing, evaluation, and decision making of the actors involved in the noise control problem. The focus will be on the decision making of the Environment Agency when setting environmental noise quality standards, and the Nagoya Association in combating Shinkansen noise pollution, since JNR's role in the process seems to be largely reactive to governmental and citizens actions. After discussing and evaluating the decision process of the Environment Agency, JNR and residents groups, methods to improve decision making for such pollution control problems will be discussed.

THE ELEMENTS OF THE NOISE POLLUTION CONTROL PROBLEM: ACTORS, THEIR OBJECTIVES AND ALTERNATIVES

Regulatory actors, objectives and alternatives

The historical, legal, and organizational framework for environmental pollution control in Japan is described elsewhere (see, for example, Environment Agency, 1976; Hashimoto 1975). Here, only the two main regulatory bodies involved in noise pollution control for Shinkansen trains will be discussed: The Environment Agency and its advisory body, the Council for Control of Environmental Pollution. The Environment Agency reports directly to the Prime Minister and the Parliament. Decisions such as environmental standard setting are made on recommendations of the Director of the Environment Agency by Cabinet orders. The legal framework for such decisions is formulated in the Basic Environmental Law for Environmental Pollution Control Measures (1967) and in special laws such as the noise regulation law of 1968.

The bulk of the information processing, evaluation and decision making on noise standards was performed by the Council for Control of Environmental Pollution. This Council is subdivided into three organizational layers: the General Meeting, the subcommittees and the expert committees. Expert committees are the research bodies which sponsor, collect and evaluate studies on special pollution problems and make recommendations about standards to the subcommittees. These recommendations have to be made purely on a scientific basis. The subcommittees are the political advisory bodies to the General Meeting and the Director of the EA. In their recommendations they have to consider the technical and scientific facts and recommendations submitted by expert committees and political, economic and other possible factors which influence standard setting. The ultimate decision is then made by the Director of the EA or by the Cabinet.

The Council began studies on noise pollution control for Shinkansen trains in 1972. Its subcommittee on noise and vibration control consisted of 15 members, including represenatives of JNR, the EA, the Institute for Public Health, the Ministry of Transportation, the Ministry of International Trade and Industry (MITI). An expert committee was formed in 1972 to study the problem of Shinkansen noise in depth. This committee had 11 members, all experts in noise pollution problems. 6 university professors, a member of the National Institute of Public Health, a member of the National Institute for Noise Problems belonged to this committee as well as experts from JNR, the automobile industry, and other industries. For implementing standards the Environment Agency has to rely on local governments and city environmental or police bodies. It should be noted that in this organizational set-up for noise pollution control decision making (as in other environmental decision making cases) no direct participation by the sufferers of pollution is provided for.

The Council is probably the regulatory actor which strongly shaped the decision on noise standard setting. To understand the objectives of the Council and the Environment Agency in setting noise standards one has to examine the Basic Law for Environmental Pollution Control of 1967. This law provides a common goal for integrated control measures based on environmental quality standards (EQS). EQS have the role of targets, desirable states to be achieved some time in the future. They are distinguished from enforcement standards (ES) which are direct regulatory tools for managing a particular pollution problem by putting enforcible limits on emmissions or total amounts of pollution. The noise regulation law of 1968 provides such enforcement standards for industrial noise, automotive noise and construction noise (see Environment Agency, 1976).

Considering environmental quality standards for Shinkansen noise first, the goals and objectives for such standards should have the character of desired states of quietness in the sense of "Maintaining and protecting human health and conserving the living environment" (Environment Agency, 1977). More concretely, noise quality standards should be set as to prevent adverse effects of Shinkansen noise on human health (damage to hearing), psychological well being (increased stress symptoms), and normal human activities (rest, conversation, etc.).

For enforcement standards economic, technical, and political objectives may be considered besides environmental objectives. Improvement of the transportation sector, governmental policy objectives of "balanced growth of the economy", agreement with international standards, reduction of complaints may be among the political and economic objectives of the regulator when setting noise enforcement standards. The economic and political objectives could possibly have an important influence on enforcement standard setting. The national government has formulated a policy to study in depth the environmental impact problems of the Shinkansen before further extensions are to be considered. In effect this means a delay of further development of several years. The major objective behind this move may not only be the concern about the environmental condition, but also concern about whether the benefits of the Shinkansen extensions would justify the enormous load on government spending. Possibly enforcement standards could thus be used to motivate restrictions on further developments of the Shinkansen.

Figure 1 is an attempt to structure the potential objectives of the Environment Agency and its Council when setting enforcement standards for Shinkansen noise. Only the environmental objectives of the upper part of Figure 1 should be considered when setting environmental quality standards. As will be shown later, the actual goals and objectives implicit in the decision making of the Environment Agency and its Council had a quite different structure, concentrating in particular on the reduction of complaints about noise.

What are the alternative regulatory means to achieve these goals? If one considers only standards as a tool, the main distinction is between EQS and ES. But there are many other dimensions on which, for example, enforcement standards can vary, including the measure on which the standard is defined, the maximum level, the monitoring and inspection procedure, area and zoning definitions, and time limits for the fulfilment of the standards. Figure 2 presents in a tree form some of the potential alternatives for noise regulation for Shinkansen trains including some alternatives to standards such as speed control and incentives. Further alternatives could include:

- o differential standards for different areas (hospitals, light residential, industrial);
- o differential target periods for achievement of standards (according to the state of construction of the Shinkansen, different areas, different base noise levels).

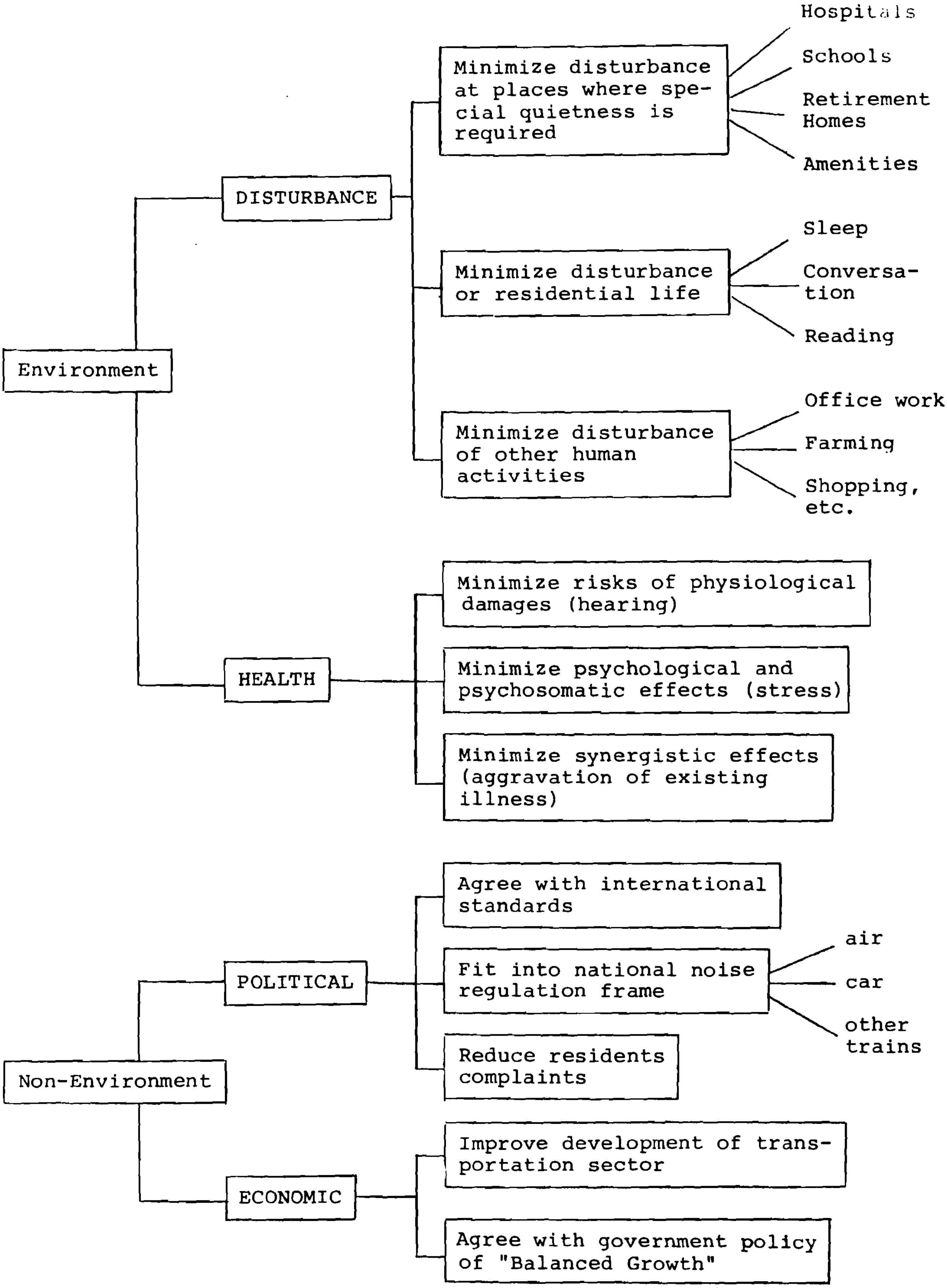
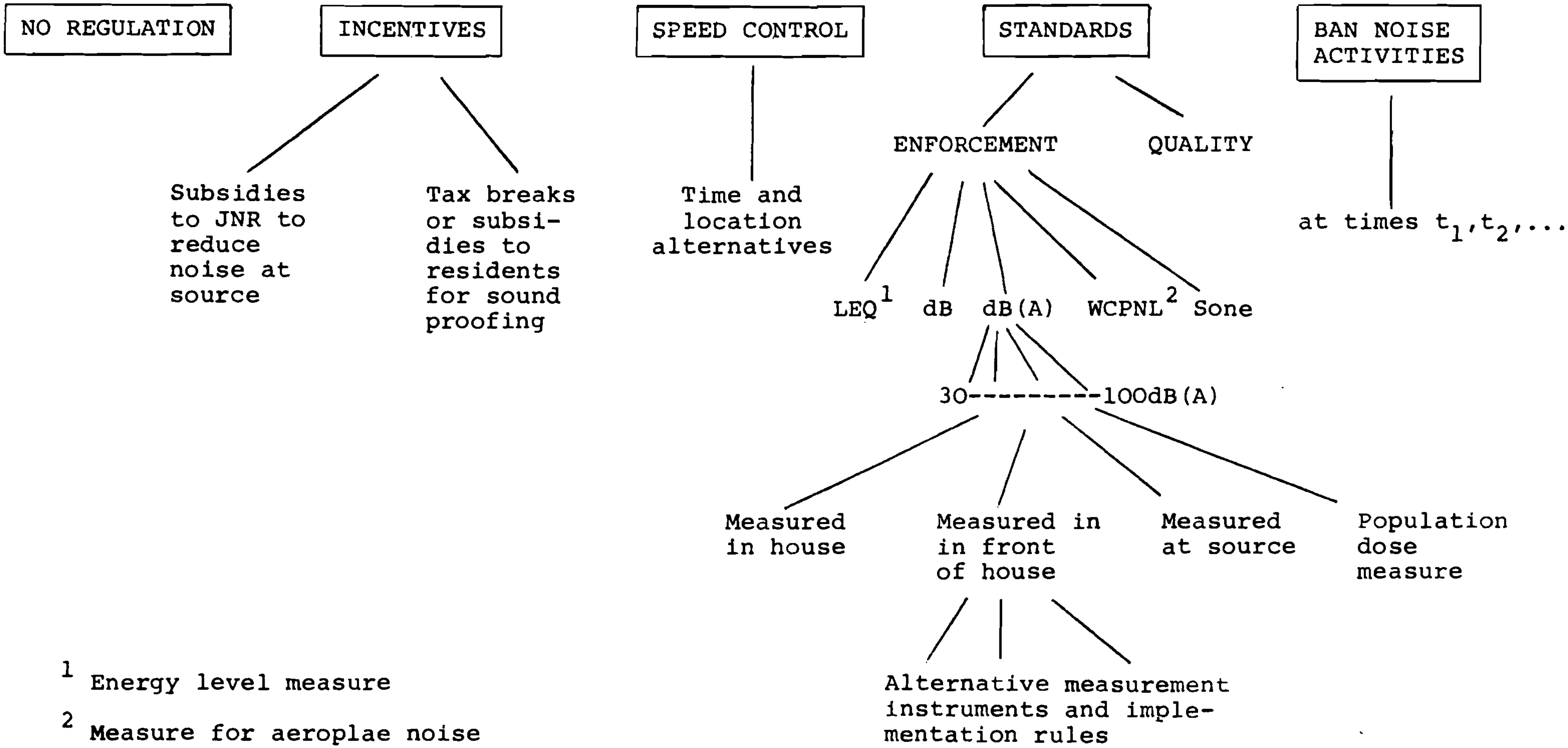


Figure 1: Regulatory Objectives for Shinkansen Noise Pollution Control



1 Energy level measure

2 Measure for aeroplae noise

Figure 2: Possible Alternatives for Shinkansen Noise Regulation

Development actors, objectives, and alternatives

Three main bodies participate in the development of the Shinkansen: the Ministry of Transportation, the Ministry of Construction, and the Japanese National Railway Corporation. The Ministry of Transportation is the main political body responsible to the Prime Minister and the Parliament in questions of planning, development, and construction of all national railways. Both JNR and the Ministry of Construction are involved in constructing railroads, with JNR usually taking the leading role. Train operations and maintenance are the task of JNR alone.

Noise pollution control for old lines as well as the development of noise abatement technology for new lines is JNR's task. Research and development on noise pollution abatement technology is done largely by JNR's Railway Technical Institute with coordinating roles taken by JNR's environmental department and its environmental committee. The environmental committee is composed of members of most JNR departments. The environmental department has 17 members which are exclusively engineers or professionals in technical services. The basis for JNR's pollution control program is described in Yorino (1977b).

The objectives of JNR in the noise pollution control decision making are structured in Figure 3. The main distinctions are between service objectives (speed, reliability, visibility), cost objectives (investment and operations), and compliance with regulation objectives.

JNR and the Ministry of Construction stressed very much the objective of maintaining a high train speed as the essential feature of the Shinkansen. Speed reduction to reduce noise was therefore never considered as a response to complaints or regulations. Although JNR officials claim that noise reduction is a genuine objective of JNR, none of JNR's decisions indicates that noise reduction measures would have been taken without regulation. Therefore, the main influence on JNR's decisions to reduce noise is by the regulatory compliance objective which stand in conflict to the cost and service objectives. It appears that JNR reacts to proposed and adopted regulations by first maximizing the service objectives and then searching for solutions which minimize cost. This interaction of objectives tends to push JNR's decision making in the direction of maintaining the status quo and reacting only slowly to outside forces.

JNR officials including those of the environmental department perceive the noise pollution problem as purely technical to be solved by its engineering branches. This fact is reflected in the proposed solutions to noise pollution control which are described in Yorino (1977b). For existing lines measures consist mainly of building sound barriers and sound proofing at the track. If these steps are not sufficient, further soundproofing is done at the receiver, and ultimately houses are being

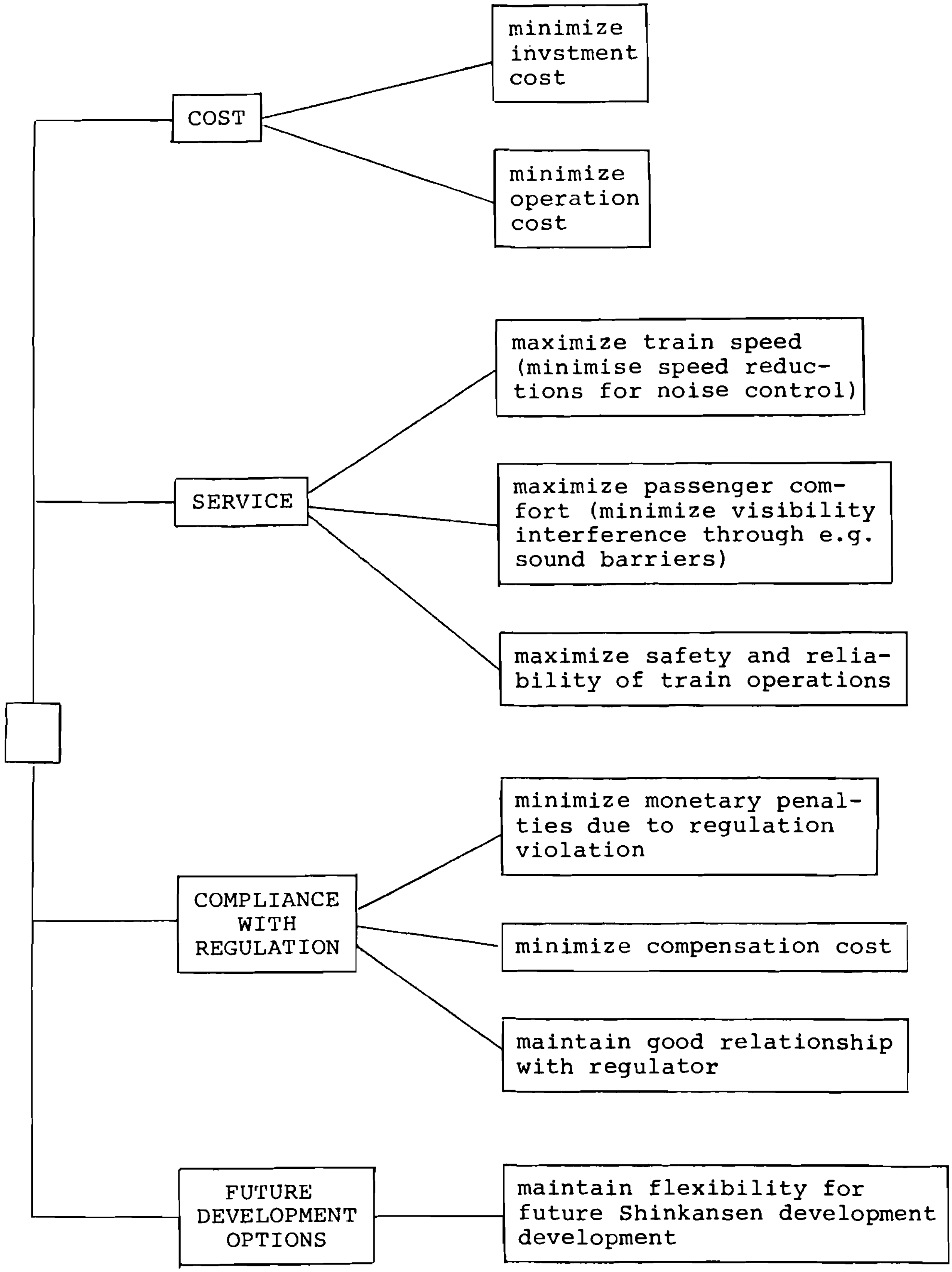


Figure 3: Potential Objectives of JNR in Decision Making for Noise Pollution Control

relocated. For new and planned lines JNR has made several research efforts to improve rolling stock, track, and structures. These developments include noise reduction technologies such as vibration proof wheels, ballast mats, slab mats, bridge hooding, etc. (see Yorino, 1977b). There are some more fundamental means to reduce noise pollution which are not seriously considered by JNR. Speed reduction in highly populated areas is the most important and simple way to reduce noise pollution. Routing is another important factor by which exposition of noise sensitive areas could be avoided. At present it appears that JNR has not made any decisions on speed reduction or routing on an environmental basis alone.

Figure 4 puts together some logical alternatives for noise pollution control (and for meeting possible standards). Further details are given in Yorino (1977b) who also evaluates the effectiveness of the various measures.

Impactee actors, objectives, and alternatives

Impactees are in the first place the residents along the lines of the Shinkansen who suffer from noise pollution. While these residents would benefit from noise regulation, the users of the Shinkansen and local businessmen and governments could possibly suffer from noise regulation. For example, if JNR should decide to slow down the train in highly populated areas, the increased travel time would reduce the benefit to the users. Also, noise abatement measures are costly and may eventually lead to higher fares. Finally, citizens actions may prevent the building of new lines from which local governments and businesses could benefit. Thus the sufferers of noise pollution and the beneficiaries of Shinkansen operations should both be included in the analysis of the impactees of noise pollution and noise pollution control measures.

The present analysis will, however, concentrate on the residents, since users, local business men and governments will have interests which are highly related to those of JNR, and they act as supporting actors of JNR rather than independent actors with own objectives and alternatives. Although in some cases local governments have entered informal bargaining procedures with JNR (for example to get a Shinkansen terminal in return for an approval of JNR's development plans) there has not been much active involvement of the users and local authorities in regulatory decision making on noise control.

Residents along the Shinkansen lines who may suffer from noise pollution can be identified in various ways:

- o those who are exposed to certain noise levels
- o those who complain
- o those who are organized in protest groups

Residents within up to 100-200 meters away from the track are probably exposed to disturbing noise levels above 70 dB. There are

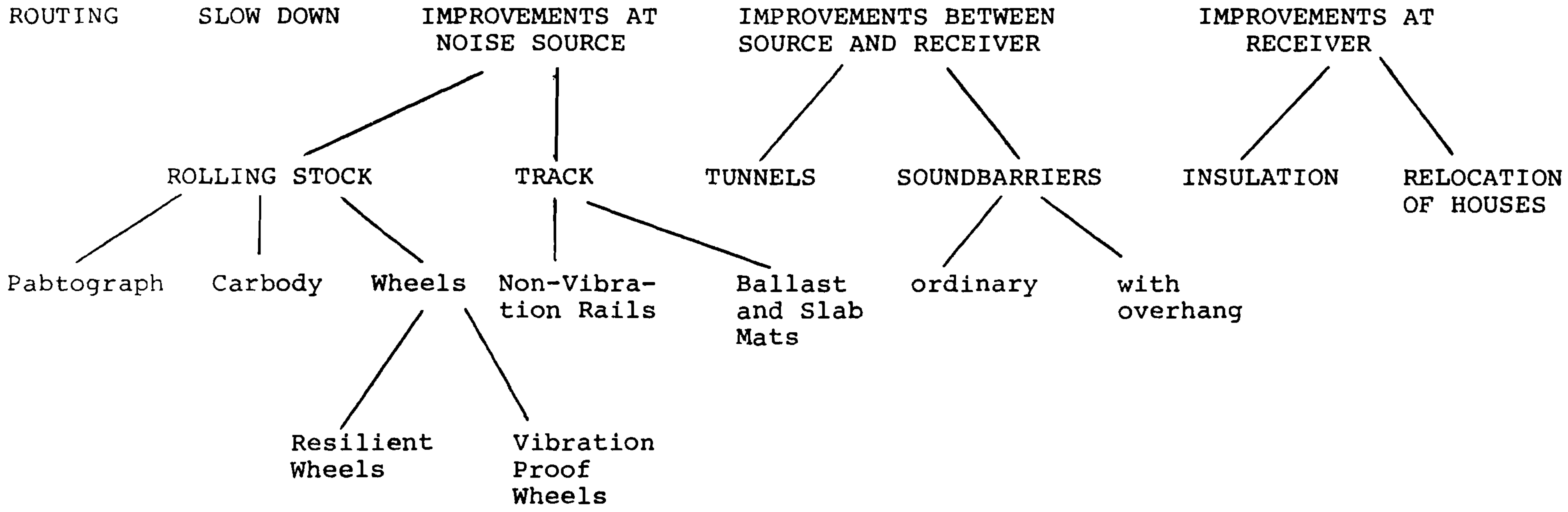


Figure 4: Alternatives for Shinkansen Noise Pollution Abatement

no precise estimates of the number of residents living within such a band along the presently existing lines, but some indications can be given. In Tokyo, for example, the Tokyo Metropolitan Environmental Institute estimates that 25,000 people in residential areas and 15,000 people in commercial areas live within a 200 meter band. The total number of "affected" residents along the Shinkansen line may well be in the hundred thousands.

The complaint statistics cited before do, of course, not reflect such large numbers. A better way to identify impactees is therefore by looking at organized groups. The most prominent group is the Nagoya Association (the "Association against Nagoya Shinkansen Public Nuisance"). This association was set up following several individual complaints, petitions, and small organizations such as the "Citizens group against Shinkansen TV interference". The association now represents 2,000 households, settled along a 7 km stretch of the Shinkansen in Nagoya.

The objectives of these residents are:

- o reduce Shinkansen noise to a non-disturbing level;
- o aid people with special diseases and illnesses
- o receive compensation for residents who have been exposed to Shinkansen noise

While the first goal is another more general formulation of the environmental objective of the Environment Agency (see Figure 1), the last two objectives are special objectives representing the self interests of the residents.

The alternative actions by which residents can move against JNR in order to achieve these objectives can be characterised by a series of escalated steps of protest:

- o complaints by individuals
- o petitions of groups to local governments, prefectures, the EA or JNR
- o organizations of various forms
- o legal litigation with alternative requirements for solving the noise problem
- o extra legal protest actions

As will be seen in the following sections, these steps have been taken by the Nagoya Association, except the most radical step of taking extra legal protest actions.

THE DECISION OF THE ENVIRONMENT AGENCY TO SET NOISE STANDARDS

While the previous sections described in a structured way the elements of the noise control problem for Shinkansen trains (actors, objectives, alternatives), the following two sections will put these elements together in a discussion of the actual information processing, evaluation, and decision making at the hand of two main issues: the setting of noise standards by the Environment Agency, and the legal action by the Nagoya residents. Stress will be put in this analysis on the priorities among objectives selected, the limited view of alternatives, the informational base to evaluate alternatives against objectives, as compared to the intentions of the decision making actors.

When the expert committee on special noise in the Council for the Control of Environmental Pollution began its investigation on Shinkansen noise its intention was to set <u>environmental quality standards</u>. In terms of the definition of <u>environmental quality standards</u> by the Environment Agency and the Basic Law on Control of Environmental Pollution this would imply that

- o only environmental objectives should be considered to evaluate standards, but not technical, economic, and political objectives;
- o alternatives which characterize enforcement standards (timing of fulfillment, area specifications) should not be part of the standards.

But in fact, as will be shown the decision process of the expert committee and the Council resulted in enforcement standards. In addition, this standard was set on the basis of information required for an environmental quality standard. This observation is not trivial since it indicates a departure from national environmental policy and implies that the information taken into account when setting the standard was insufficient.

As a first step the expert committee reviewed the present state of Shinkansen noise pollution. These results are documented and discussed, for example, in Environmental Agency (1973), Hashimoto (1975), and Yorino (1977a and b). The general outcome of these studies was that Shinkansen noise levels range between 75 dB and 120 dB. At about 25 miles away from the source (see Table 2) JNR claims that the maximum noise level is 110 dB.

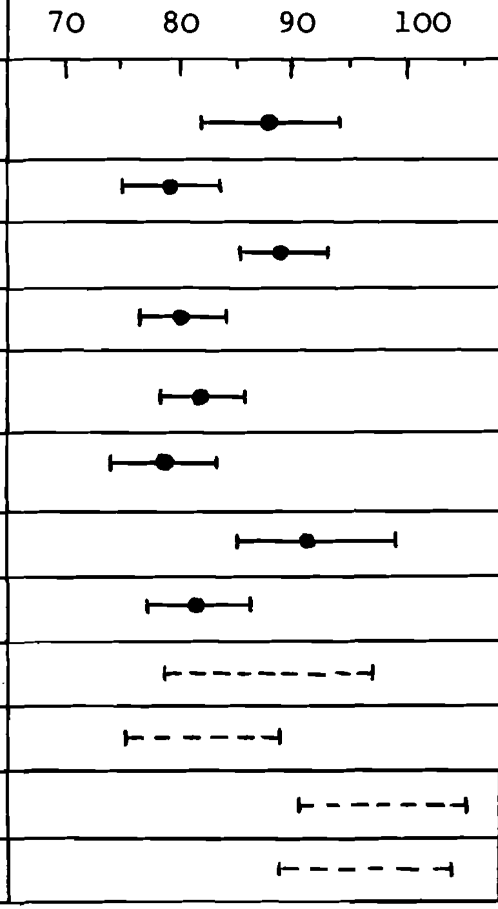
Next, the expert committee issued studies on the effects of noise. The primary intention of these studies was to establish the relationship between noise and complaints <u>not</u> to identify the psychological and physiological effects of the Shinkansen noise. One could of course argue that complaints are an indicator of more indirect and less observable effects, and that the absence of complaints would indicate an absence of effects. However, no such arguments were made by the decision makers involved in the process of standard setting. Rather the noise-complaint relationship was considered a main driving force itself for the stadard setting decision.

Several studies on noise versus complaints produced results which are exemplified in Figure 5. In general there appears to be a relatively slow rise in complaints between 50-65 dB. From

TABLE 2: SHINKANSEN NOISE LEVELS

Source: Hashimoto, 1975

STRUCTURE	BALLAST	SIDE WALL	MEASUREMENT POINT	NOISE LEVEL dB(A)	RAILWAY
Banking	Crushed Stone Ballast	Not Installed	Measured Track		Tokaido Line
			Opposite Track		
Elevated Bridge	Crushed Stone Ballast	Not Installed	Measured Track		Tokaido Line
			Opposite Track		
Elevated Bridge	Crushed Stone Ballast	Side wall of 1.9 m high is installed	Measured Track		Sanyo Line
			Opposite Track		
PC Girder	Crushed Stone Ballast	Not Installed	Measured Track		Tokaido Line
			Opposite Track		
Iron Girder	Crushed Stone Ballast	Not Installed	Measured Track		Tokaido Line
			Opposite Track		
Iron Girder	No Ballast	Not Installed	Measured Track		Tokaido Line
			Opposite Track		



1. Measurement was made at a point apart from the center of the tracks by 25 meters and at a height of 1.2. meters from the ground level.
2. and indicate the 90% range of measurement values and the average value respectively. And indicates the whole range of measurement values because the measurement points were so few.

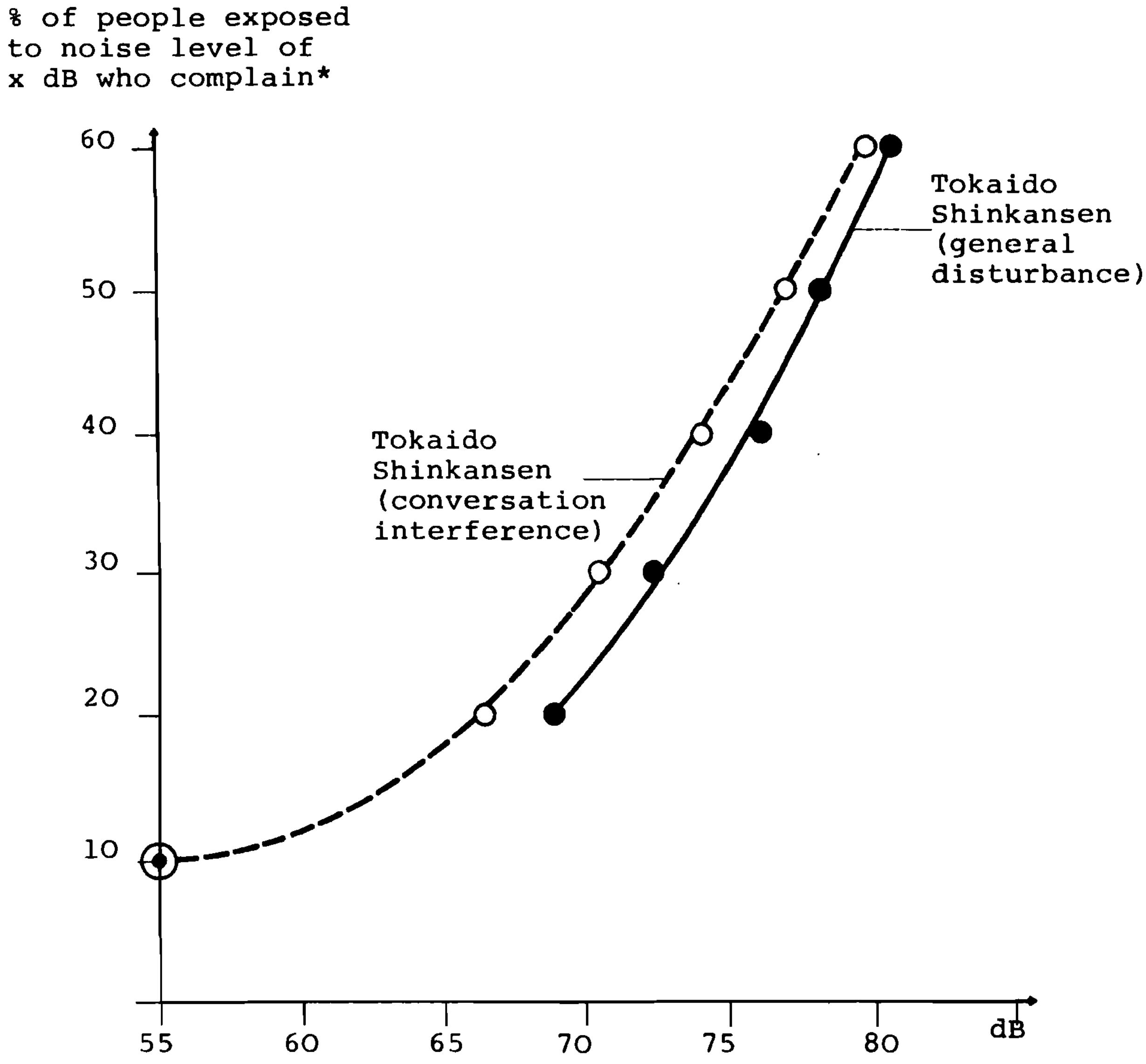


Figure 5: Complaints versus Noise Level

*Source: Environment Agency (1975)
"complaint" is defined as one of the following responses:

"occasionally experienced disturbance"

"rather frequently experienced disturbance"

"frequently experienced disturbance"

65 dB onwards complaints begin to rise rapidly. If one wanted to set standards on the basis of such noise-complaint relationships, one would attempt to either set the standard such that no complaints occur, or such that complaints are relatively stable below that standard, arguing for some sort of a threshhold effect. The threshhold idea was apparently assumed by most of the experts involved in the standard setting, in the sense that they believed that there is a "normal" background level of complaints which has little to do with actual disturbances. With such a position the problem naturally shifts from identifying a noise level with zero complaints to identifying a noise level at which complaints just reach "background" rates.

No studies on economic, technical, or other questions were performed or issued by the expert committee. JNR had done some such studies (see Yorino 1977a and b), particularly on the technical feasibility and effectiveness of noise pollution abatement, but they were not considered in the deliberations on standards. This "neglect" is, of course, in line with the intention of environmental quality standards, although such studies would have been necessary to evaluate enforcement standards. Cost estimates were made by JNR, although they have not been made public. In individual discussions JNR experts estimated that a reduction of the noise level down to 80 dB would increase investment cost by 9% and a further reduction to 70 dB would increase investment cost by 13-14% (see Figure 6).

Studies on the effect of train speed on noise levels existed but were not explicitly considered by the expert committee. Figure 7 exemplifies the basic result of such studies: There is an almost linear drop in noise between 200 and 100 km/h and a small rise at levels between 50-60 km/h. At least this is the intuitive impression from looking at the data. The interpretation of JNR and other experts of this data is different: on the basis of a priori considerations about the relationship between speed and noise, these experts hypothesised that the relationship between noise and speed must be logarithmic. A least square fit would then lead to the curves shown in Figure 7. Such a logarithmic relationship was also a main argument for JNR to refuse slowing down as a tool for reducing noise: JNR claims that the noise reduction would be only slight between 200 and 100 km. If one looks at the fitted curve, this is true. If one looks at the data points it is not.

Having reported now on quite a number of available data which have not been considered by the expert committee it seems surprising that standards were set purely on the basis of the noise-complaint relationship. In 1975 the expert committee recommended (and the Council accepted) to set a standard of 70 dB for the mainly residential areas, and of 75 dB for mixed residential and commercial areas. The rationale behind this recommendation was that at a noise level between 70 and 75 dB approximately 30% complaints was achieved. The study group maintained that only substantial reductions of the noise level would lead

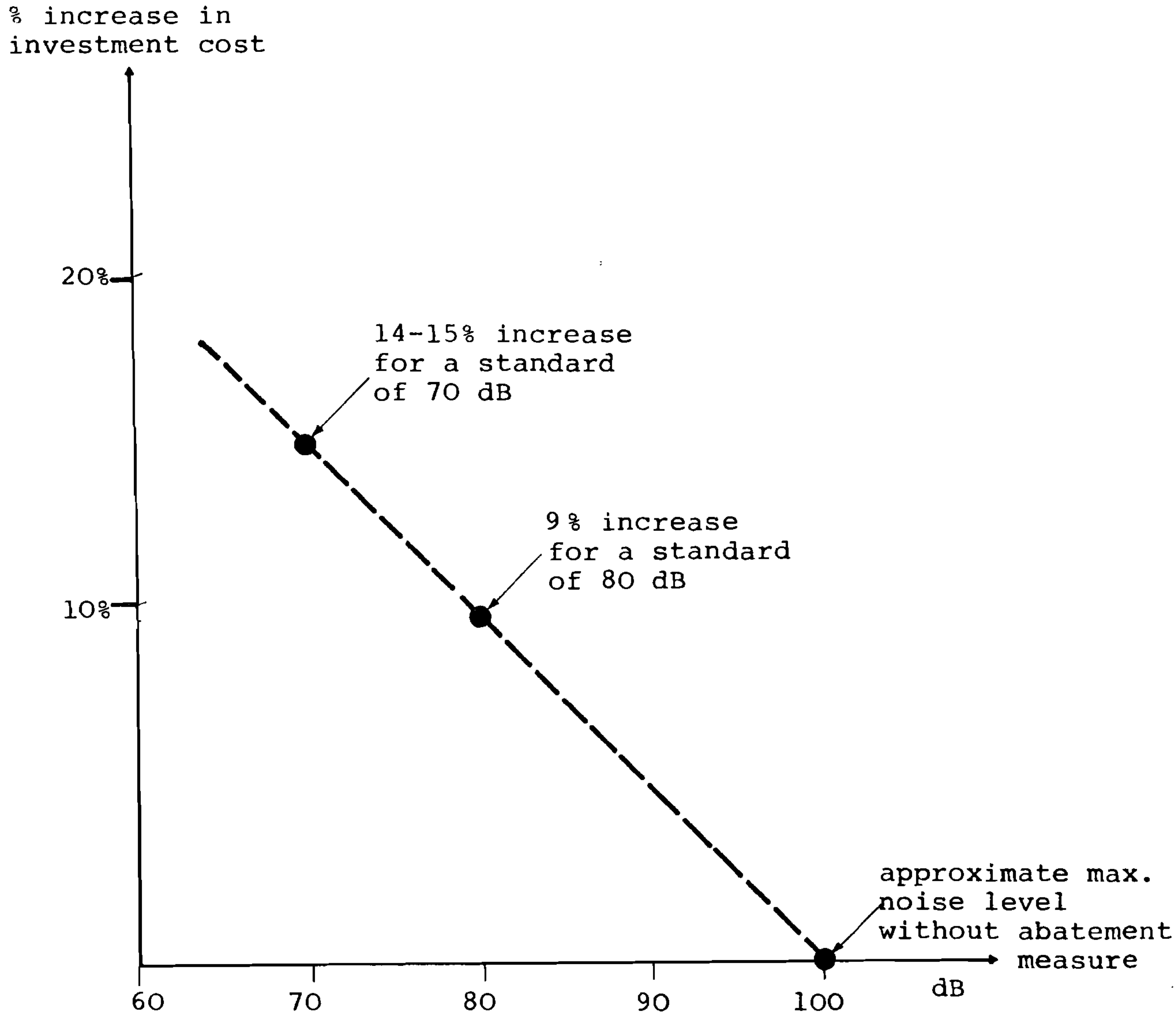


Figure 6: Increase in Investment Cost versus Noise Level* (Sanyo Line)

*YORINO, private communication

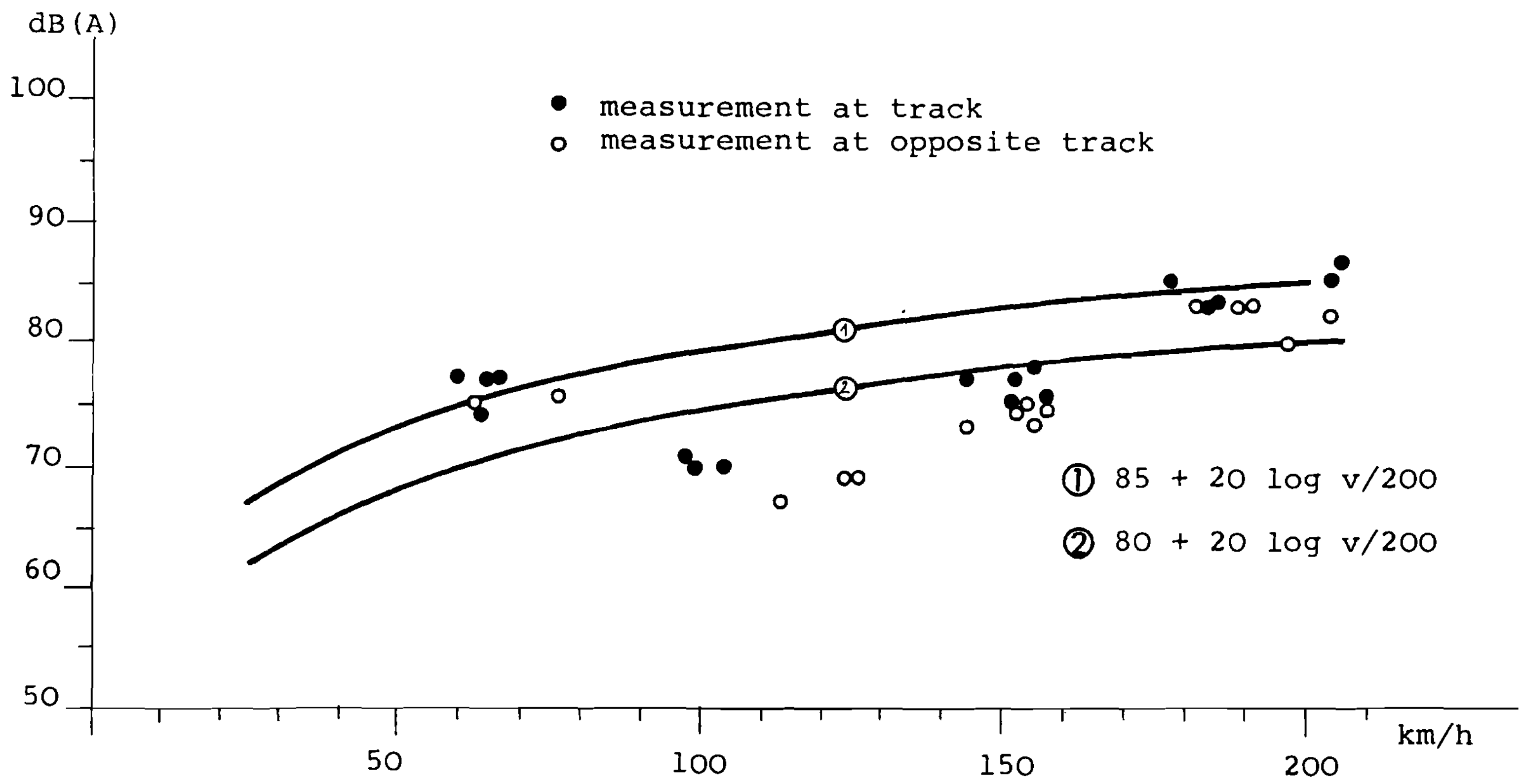


Figure 7: Train Speed versus Noise

Source: Environment Agency, 1977

to any further reductions of the complaints. On the basis of Figure 5 such conclusions appear rather bold. In addition, one would have to ask, whether 30% complaints are actually acceptable from an environmental quality standards point of view.

But further recommendations by the expert committee dispelled all thoughts that quality standard were being set. These recommendations included area specifications for the implementation of the standards as well as time limits for JNR to comply. So in fact they were turned into enforcement standards, although the literature still calls them environmental quality standards. (see Hashimoto, 1975). The Environment Agency accepted the recommendations by its council and the standards went into effect in September 1975.

JNR, being represented in the expert committee and in the subcommittee had suggested 80 dB as an alternative to the standards which were finally accepted. These suggestions were based partly on economic considerations, partly on an evaluation of recent technical studies. JNR especially doubted (and still doubts) that 70-75 dB can be achieved with present technology on "noisy" structures such as steel girder bridges. JNR had, as indicated before, taken a firm stand on not slowing down as a means of reducing noise and apparently such a measure was never a serious issue in the expert committee.

Since the setting of the standards JNR has made major efforts to meet the time limits for the standards in areas with noise levels higher than 80 dB. As a first step a string of sound barrier walls were built. In 1974 these efforts cost JNR, for example 15.1 billion Yen. They reduced the noise level by about 5 dB and JNR estimates that all existing lines now meet a level of 80 dB or less except for some steel girder bridges. As to the future JNR is uncertain whether it will be able to meet the standards in the prescribed period of time.

As is readily discernible there were no direct inputs or any participation by the impacted residents along the Shinkansen lines during this whole process of standard setting. Residents have in the meantime fought a quite different battle of protest and lawsuits. This process will be discussed next.

THE DECISION PROCESS OF THE NAGOYA ASSOCIATION TO COMBAT SHINKANSEN NOISE

This section is largely based on a discussion with two lawyers representing the Nagoya Association against Shinkansen Public Nuisance. It is a descriptive account of the main steps in the battle between the Association and JNR in which the residents tried to achieve their objectives described in Figure 4.

The first complaints by Nagoya citizens reached the Nagoya city council in 1964 shortly after the beginning of operations of the Tokaido line. At issue was a 7 km stretch which is elevated and runs partly over steel girder bridges. In a first significant move, 110 Nagoya residents jointly delivered a complaint and a petition for support to the Nagoya City Council. In 1967 the Nagoya Council, after performing own noise measurement and confirming high noise levels, asked JNR to take measures on noise pollution abatement. JNR responded in March 1970 by announcing that sound barrier walls would be installed at three points along the controversial 7 km stretch. Meanwhile several small residents groups organized into associations to fight Shinkansen noise.

In 1971 these organizations united as the "Association against Nagoya Shinkansen Public Nuisance" which comprised about 2000 households, and is subdivided into seven neighbourhood associations. At the same time JNR began installing sound barrier walls at several points, mainly in hospitals and school areas.

Between 1972 and 1974 several meetings between the Association, JNR, the Mayor of Nagoya, and prefectural officers as well as the Environment Agency took place. The initial request of the citizens was that JNR should increase sound barriers over the whole 7 km length. Later the Association began to take a firmer stand:

- o the Shinkansen should be slowed down in order to bring noise and vibration down to non-disturbing levels;

- o special measures should be taken by JNR to aid ill and old people.

JNR's offer in turn was to take steps to reduce the noise down to 80 dB, but flatly refused to slow down. 80 dB was not considered acceptable by the residents. JNR agreed to examine the special cases of sick and old people, but up to 1977 no such case was resolved.

The local city and prefectural governments took action at the end of 1972 by formally supporting the Association and requesting JNR to slow down in the 7 km stretch. In August 1973, finally, a group of 20 lawyers was set up by the Association to begin legal action against JNR on behalf of 570 residents. The case requested specific measures to be taken, once more raising the stakes in the fight between JNR and the residents:

- o noise levels should not exceed 65 dB between 7 a.m. and 9 p.m.;

- o noise levels should not exceed 55 dB at other operating times;

- o the area to be considered was 100 m of both sides of the track along the 7 km stretch;
- o for each plaintiff JNR should pay 1 Mio Yen compensation;
- o JNR was to carry the cost of the court case.

New residents could join the case provided they lived within 100 m of the track and joined within 3 years after the case was opened. Since the lawyers worked on a no-fee basis costs of joining were small, but there was no dramatic increase in the number of plaintiffs in recent years.

The standards which the Association requested were based on similar material as the one that the Environment Agency used for their standard setting. No independent research effort was made to substantiate the claim of the residents that levels above 55 dB (or 65 dB during the day) were disturbing. The setting of noise standards in 1975 therefore was a major event in the court case. The Association did not believe that 70-75 dB would stop the disturbance and pursued the case after standards were set. However, there is now some pessimism with respect to the outcome of the case since the lawyers believe that the judge will orient himself towards the EA material on quality standards.

JNR's response to the case in its first court session in 1974 was to deny the request on the grounds that they were "totally unreasonable". This tough stand was somewhat toned down in the following months. JNR argued that 65 dB could not be achieved with presently available technology without a slow down and thus a substantial delay of trains (in fact, the delay at the 7 km stretch from slowing down from 200 to 100 km/h would be less than 5 minutes. However, by slowing down an important precedent would be set which is in conflict with JNR's objectives). JNR proposed to remove and relocate all houses within 20 m of the line. After the standards were set, JNR proposed (as the standards require) to implement 80 dB standards within three years.

The case now is still pending. A judgement is not expected before the summer of 1978. But the movement of residents, and local governments has already shown some effects which were not expected. One of the two labour unions of JNR expressed its support for the Association and asked train operators who are members of this union to slow down their trains to 100 km/h on the controversial stretch. The result was that 50% of the trains drove at a much slower speed and reduced the noise level in spite of JNR's tactics.

The decision process shows that the residents had quite different objectives and action alternatives in mind than the EA and that they went head on against JNR after initial requests failed to produce any results. It also demonstrates the gradual escalation of both objectives (from sound barrier improvements to standards and compensation) and of tools (from complaints to legal litigation). JNR's role was as reactive in this process as it was in the process of standards setting. It appears that this strategy was successful in saving time, but it clearly raised the stakes.

DISCUSSION AND EVALUATION: TOWARDS MODELLING STANDARDS SETTING DECISIONS

The analysis of actors, objectives, and alternatives in noise pollution control was meant to demonstrate the wide span of elements which could or should have entered into the decision processes. Looking at actual information collection evaluation, and decision making of the Environmental Agency, the Japanese National Railway Corporation, and the Nagoya Association it becomes obvious that not all these elements were considered, that information was neglected, that evaluations were partially biased, and that the interactions between the three decision making groups was limited.

In this discussion the decision processes of the three groups will first be analysed and evaluated in terms of their own intentions entering into the decision making. The discussion will conclude with some suggestions how decision and game theoretic models may aid regulators and other decision makers in pollution control decision problems.

The organizational set up for standard setting decisions in the Japanese Environment Agency has some unique features which reflect the original intention of the EA to set Environmental quality standards rather than enforcement standards:

- o clear separation of expert analysis and expert judgement for questions of health, and environment on the one hand, and the political decision process on the other;
- o main location of the decision making in the expert bodies of the Council (expert committee, subcommittee) with consensus forming and checking functions located at higher levels (EA, Cabinet);
- o participation of many experts from industry and government, but exclusion of impactees.

For environmental quality standards this organizational set up seems to work rather well: the main task is for experts to identify risks and hazards, and to recommend acceptable pollution levels from an environmental and health point of view. All other decision layers would then judge the expert recommendation mainly on the basis of political feasibility, and perhaps on some rather general economic and technical considerations. For enforcement standard setting this process could also work well, if the expert committee would consider all scientific aspects of the standard setting problem including economic and technical aspects. Given this information on environment, health, economic, and technical issues the higher decision making layers could then make trade-offs, add political judgements and make final decisions.

However, as was mentioned before, the main problem with the noise standard setting decision process of the Environment Agency was that studies and evaluations were done as if environmental quality standards were set, while in fact the outcome was enforcement standard. What would have been an appropriate analysis for quality standards, falls short for enforcement standards:

- o only limited alternatives were considered (no consideration of the effects of slowing down, zoning, house removal, incentives, etc.);
- o information on health, costs, broader economic and transportation system impacts, technical aspects etc. were not considered;
- o the standards were evaluated on the basis of very limited information on noise-complaint relationships and a relatively arbitary standard was set.

These shortcomings are to some degree recognised by the decision makers in the EA and JNR. Yorino, for example, states that "this ambiguity (about standards as a target or a tool)* seems to have been not properly cleared at the discussion of environmental quality standards for the Shinkansen noise. It also seems that ... (questions of) ..., the impacts of the national economy in the light of the prevailing economic situation, how far other kinds of noise are controlled, how about the living standard of the nation, etc. all not having been properly studied." (1977b, 18).

JNR's decision making in the standard setting process and in its negotiations with the Nagoya Association was largely reactive. It does quite well reflect JNR's desire to maintain the staus quo and not to forclose any future development options. JNR's reactive decision making was characterized by the following features:

- o the noise pollution control problem was seen as a problem of inventing technical solutions to respond to regulations, rather than as a problem of developing plans and procedures for abating noise pollution;
- o routing and slowing down were not considered as noise control measures;

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* Insertion by the author

- o slowdown effects on noise reduction were played down in spite of scientific evidence;
- o comfort, speed, reliability, and national economic importance of Shinkansen were stressed;
- o JNR's alternative suggestions for standards and noise abatement were often made to gain more time.

Once standards were set in 1975, however, JNR acted quickly to implement the measures necessary to comply and to develop research and technical skills to meet the future requirements. Yet this response is still purely technical and no attempts have been made to develop a comprehensive environmental management program for the Shinkansen.

The Nagoya Association probably had the most difficult role in the decision process on Shinkansen noise pollution control. The main obstacle for the Association was the lack of a direct input into governmental decision making on noise standards. It therefore had to fight an independent battle and it is still highly uncertain whether it will meet its objectives. The Association did manage to enlist support from local and prefectural governments and labour unions. It did attempt to interact with a broad array of actors, but it apparantly was not very effective in its individual bargaining with the EA and JNR. Perhaps the Association could have obtained some improvements early in the 1970s (suggested by JNR were sound barriers on the whole stretch) but the residents chose to escalate and raise the stakes. The most problematic point in the Associations decision making is the request for fixed standards in the law suit. Since the Association does not have its own research material to back these standards, the judge may well follow the EA arguments and make the official standards a basis for the case. In retrospect it would have been wiser to leave the burden of proof that the Shinkansen does <u>not</u> disturb to JNR rather than offering an opportunity for being challenged to prove that 65 or 55 dB are non-disturbing levels.

While each actor group had its own decision making difficulties and shortcomings, the major problem with the decision making on the Shinkansen noise appears to be the lack of comprehensiveness in the consideration of objectives and alternatives, and the lack of interaction between the decision makers involved. Both shortcomings are clearly demonstrated in the difference between what could have entered the decision making (alternatives, objectives, information) and what actually did enter it. One may conceive of various ways to improve the decision making for future standard setting cases, including institutional, procedural, and methodological innovation. Public participation, science courts, decision and game theoretic methodologies could be considered. Possibilities of improving the institutional aspects of standard setting and regulation are, for example, discussed in reports by the Academy of Sciences of the US (1975) to the National Research Council (1977).

Formal approaches to improve standard setting decisions are presented for example for the case of water pollution in Dorfman and Jacoby (1970), for noise in Loucks (1977). A decision theoretic model to improve standard setting and pollution control decisions has been developed and applied to chronic oil discharge standards (v. Winterfeldt, 1978a and b). It begins where this paper leaves off: by hierarchical structuring of objectives and alternatives for the three generic decision making groups, the regulator, the developer, and the impactee group. Stress is put in this structuring on breadth and comprehensiveness of objectives and alternatives.

In the quantification part the uncertainties and values of the three decision making groups are then quantified by judgemental probability distributions and utility functions. The main outputs of the model are the optimal (in the decision model sense) responses of the developer and the impactee to a given regulation, and an evaluation of regulation, the developer's response, and the impactees' response in terms of their own objectives.

The application of this model to chronic oil discharge standard setting showed that there can be some benefit from structuring the problem rigoroursly: new alternatives are discovered, alternatives included for reference provide interesting cornerstones, attempts to operationalise objectives provide new ideas on the structure of goals. Also, several insights can be gained from quantification and model run: sensitive decision points can be identified, dominated strategies can be uncovered.

But perhaps the most useful application of decision theoretic models and ideas is their use as tools for creating a more rational dialogue between different decision making units. A decision model can aid regulators, developers and impactees to communicate their values and uncertainties in a more precise way, to identify where the sources of conflict are, and to assess the consequences of changing model parameters according to the specifications of the actors involved.

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II.

DECISION AND GAME THEORETIC MODELS FOR STANDARD SETTING

A DECISION AIDING SYSTEM FOR IMPROVING THE ENVIRONMENTAL STANDARD SETTING PROCESS*

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Project, "Procedures for the Establishment of Standards"

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A DECISION AIDING SYSTEM FOR IMPROVING THE ENVIRONMENTAL STANDARD SETTING PROCESS

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THE STANDARD SETTING PROBLEM

The increased public awareness of the risks and hazards of modern technologies has led to a variety of regulatory actions to reduce detrimental effects of development and production processes on human health, safety and psychological well being. These regulatory actions include banning of certain industrial activities, emission taxes, standards and equipment specifications. Among these, standard setting has evolved as one of the most widely used regulatory tools to reduce environmental pollution.

The regulatory decisions in standard setting include a choice of the measure on which a standard is to be defined (e.g., the average concentration of a chemical in a defined medium), the level of the standards (the maximum permissible amount), the monitoring and inspection procedure, and possible sanctions for violations. Regulators and scientists who have been involved in such decision processes admit that the task is exceedingly difficult and that the results of regulatory deliberations often lack a sound scientific basis.

The main factors that complicate standard setting decisions are the following:

- There usually exist substantial uncertainty about the effects of pollutants on human well being. Experts differ widely in their judgements of pollution effects.
- Standards have to be set in the light of conflicting objectives such as environmental, engineering and economic objectives. Many objectives are hard to quantify and some can even be difficult to express precisely.
- The regulator is usually not a single decision maker, but various administrative units and experts interact in standard setting. In addition, different groups with often conflicting interests are affected by regulatory measures.
- Effects of pollution and regulation are distributed unevenly over time. Crucial trade-offs have to be made between costs and benefits today, and costs and benefits in the future.

To overcome some of these difficulties, several formal techniques and analytical approaches have been suggested along which the regulator could organize his information collection and evaluation tasks. Several authors have suggested and applied cost-benefit analysis to standard setting and regulation (1,2,3). These techniques are often criticised for being too rigid and formalized and for being unable to match the complexity of the standard setting problem (4). Less formal policy analyses of standard setting problems on the other hand (5,6), can provide interesting insights in the general nature of the standard setting decision problem but are often of little help for solving the very practical problems of the regulator.

This paper presents a decision aiding system for improving the standard setting process which settles somewhere between rigid cost-benefit analysis and policy analysis. The decision aiding system takes the regulator and his standard setting problem as a starting point. The system connects several decision theoretic tools and techniques in a decision theoretic model which allows the regulator to structure the standard setting problem, to quantify uncertainties, intangibles and values of the groups involved in standard setting and to identify an acceptable solution. The decision theoretic model is not a usual optimization model in the sense that some real world system is modelled and optimized. Rather it is a cognitive roadmap along which the regulator and his experts can organize their thoughts, preferences and information.

THE GENERAL DECISION THEORETIC MODEL

Figure 1 presents in qualitative terms the tasks that a decision maker should go through before making a decision (7,8). First, the decision maker has to identify the decision unit for which he assumed the decision making responsibility. Second, the decision maker has to define the goals and objectives of this unit. Third, the decision maker has to define the alternative courses of action by which these objectives can be achieved, the possible external events and actions that may influence his actions and the information sources available to him. For each alternative, external event and information he should then assess the probable consequences. Finally, he has to evaluate these consequences against his objectives, and on the basis of these assessments, he can evaluate his action alternatives. Then he can decide either to collect more information to improve the precision of his prediction or to directly select the best action alternative.

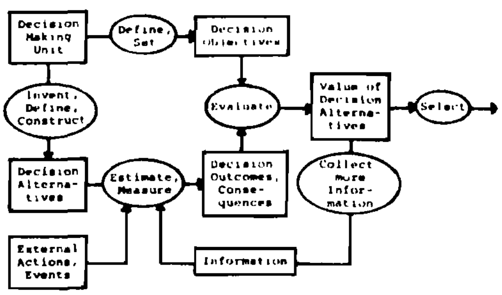

Fig. 1. Schematic representation of decision making tasks

These qualitative considerations are formalized in Fig. 2. In the ellipses the decision theoretic tools are listed which can aid the decision maker in the respective tasks:

- Goals trees structure general supergoals and operational objectives (attributes) in a hierarchical manner (9).
- Decision trees map out the logical decision alternatives and link them to external actions, events and information variables (10).
- Judgemental probability methods and models aid the decision maker and his experts to quantify their uncertainties (11).
- Multiattribute utility models and techniques aid the regulator to evaluate consequences which are uncertain and vary on several value relevant attributes (12).

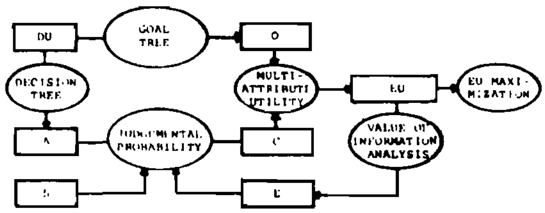

Fig. 2. Formal representation of decision making tasks

Notation:

DU: Decision Unit
A : Decision Alternatives ($a \in A$);
O : Set of Objectives ($O_i \in O$; $i = 1,2,\ldots,n$);
C : Value relevant consequences ($C \subset \mathbb{R}^n$; $c \in C$);
S : External actions or events ($S_j \in S$; $j = 1,2,\ldots,m$);
E : Information sources ($e \in E$);
EU: Expected utility ($EU(a,e)$).

The formalized output of the decision tree and judgemental probability analysis is a probability distribution p over the external actions or events S, which assigns a probability $p(S_j)$ to each element in S. The model assumes that the events and external actions are mutually exclusive and exhaustive. The other output of these estimation tasks are probability density functions $f(\underline{c}|a,S_j,e)$ defined on C, which are a quantitative expression of the decision maker's remaining uncertainty about consequences given an act, external event and his information.

The output of the goal tree and multiattribute utility analysis is a v. Neumann and Morgenstern utility function (13) $u(\underline{c})$ which is defined over consequences C. v. Neumann and Morgenstern utility functions have the property that they preserve the order of the decision maker's preferences over C and that their expectation preserves the preferences over probability density functions f defined over C. Given $u(\underline{c})$, p, and $f(\underline{c}|a,S_j,e)$ the expected utility of an action a given e can be computed :

$$EU(a,e) = \iint_C \Big[\sum_{j=1}^{m} p(S_j|e) \cdot f(\underline{c}|a,S_j,e) \Big] u(\underline{c})\, d\underline{c}. \qquad (1)$$

This expected utility is then an appropriate index for ordering actions a based on information e.

Given these functions the expected utility maximization model allows the decision maker to

- select the action a* with the highest expected utility based on information e:

$$EU(a^*,e) = \max_A EU(a,e) \qquad (2)$$

- perform a value of information analysis by recomputing the expected utility for different information sources.

Particularly the expected value of perfect information, and the expected value of sample information (14) can be computed, if the cost of information is included as an attribute in the consequence space C. Further sensitivities of the model can be analysed by changing the utility function u or the probability density functions p and f.

THE REGULATOR-DEVELOPER-IMPACTEE MODEL

In standard setting the decision making problem is complicated by the fact that several decision making units are involved in the decision making process. The general decision theoretic model above is adapted to these circumstances by postulating that in standard setting three main decision units interact: *the regulator unit*, *the developer unit*, and *the impactee unit*. The regulator unit is defined by the administrative unit which has the responsibility for setting a standard. The developer unit is defined as all those decision units whose production and operation decisions are restricted by the regulation. The impactee unit is defined as those groups or organizations who are potential "sufferers" from the respective development actions. The decision problem of the regulator is to set a standard which balances the opportunity costs imposed on the developer against the potential benefits for the impactees.

These three decision making units interact in standard setting as presented in Fig. 3. Each unit has its own alternatives, objectives, consequences, estimation and evaluation tasks which follow the schematic outline of the previous section. The linkages are created through the external event boxes of Figs. 1 and 2. The model does not spécify external events for the regulator. But regulatory action creates the possibility of external events for the developer. Given a standard, monitoring and inspection procedure and sanctions for violations, the developer faces the possibility that he will incur sanctions if violations are detected, unless he introduces measures to reduce possible pollution level. The model assumes that the external event "detection of violations" together with the resulting sanctions change the developer's consequence distributions and thus his evaluation of development activities.

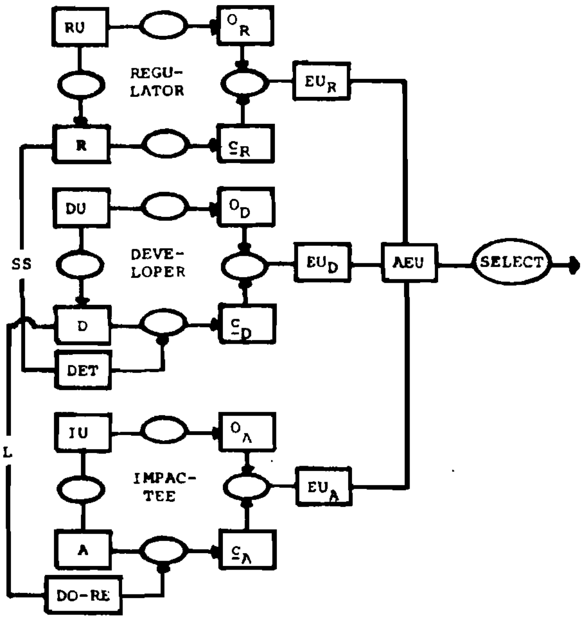

Fig. 3. The regulator-developer-impactee model

Notation:

RU,DU,IU	:	Regulator, Developer, Impactee decision unit;
R,D,A,	:	Alternatives of regulator, developer and impactee (typical elements: r,d,a,);
O_R, O_D, O_A	:	Objectives of regulator, developer and impactee;
C_R, C_D, C_A	:	Consequence spaces (typical elements: $\underline{c}_R, \underline{c}_D, \underline{c}_A$);
EU_R, EU_D, EU_A	:	Expected utilities of regulator, developer and impactee;
AEU	:	Total expected utility of regulation;
SS	:	Possible sanctions;
DET	:	Detection submodule (see text);
L	:	Possible pollution levels
DO-RE	:	Dose-Response submodule (see text).

The linkage between the developer unit and the impactee unit is established through levels L of pollution resulting from production or operations which change the consequences of the impactees. This view is usually represented in the dose-response models of environmental pollution and resulting health or other consequences. In the model this dose-response concept is formalized in the changed consequence distribution for impactees.

For a mathematical formulation of the model, the following further definitions are made:

F_R, F_D, F_A: Probability density functions over the respective consequence spaces C_R, C_D, C_A $(f_R \varepsilon F_R;\ f_D \varepsilon F_D;\ f_A \varepsilon F_A)$;

u_R, u_D, u_A: v. Neumann and Morgenstern functions over the respective consequence spaces C_R, C_D, C_A;

SL : set of standard dimensions and levels (slεSL);

SM : set of monitoring and inspection procedures (smεSM);

SS : set of sanctions for violations (ssεSS);

TP : set of treatment processes to reduce pollution (tpεTP);

TE : set of treatment equipments (teεTE);

TO : set of treatment operations (toεTO);

E_R, E_D, E_A: information sources of the regulator, developer and impactee $(e_R \varepsilon E_R;\ e_D \varepsilon E_D;\ e_A \varepsilon E_A)$;

n_q : number of detections of cases in which the actual level of pollution l exceeds the standard sl during the time interval of interest T $(n_q = 0,1,\dots,N)$;

$p(n_q)$: probability distribution over n_q;

$f(l|d)$: probability density function over levels, given d;

R and D are defined as product sets:

$$R = SL \times SM \times SS \tag{3}$$

$$D = TP \times TE \times TO \tag{4}$$

Based on these definitions, the standard setting model postulates the following decompositions of the probability density functions f_R, f_D, and f_A:

$$f_R(\underline{c}_R|r,d,a,e_R) = f_R(\underline{c}_R|r,e_R) \tag{5}$$

$$f_D(\underline{c}_D|r,d,a,e_D) = \sum_{n_q=0}^{N} p(n_q|d,sl,sm,e_D) \cdot f_D(\underline{c}_D|d,ss,n_q,e_D) \tag{6}$$

$$f_A(\underline{c}_A|r,d,a,e_A) = \int_L f(l|d,e_A) \cdot f_A(\underline{c}_A|a,l,e_A)\,dl \tag{7}$$

The model assumes further that the regulator can construct the functions u_R, u_D, u_A and the probability density functions f_R, f_D, and f_A (as specified earlier) on the basis of his information e_R. The regulator can then determine his expected utility EU_R and his guesses of the expected utility EU_D and EU_A as follows:

$$EU_R(r,e_R) = \int\!\!\int_{C_R} f(\underline{c}_R|r,e_R)\,u_R(\underline{c}_R)\,d\underline{c}_R \tag{8}$$

$$EU_D(d,r,e_R) = \int\!\!\int_{C_D} [\sum_{n_q=0}^{N} p(n_q|d,sl,sm,e_R) \cdot f_D(\underline{c}_D|d,ss,n_q,e_R)] \cdot u_D(\underline{c}_D)\,d\underline{c}_D \tag{9}$$

$$EU_A(a,d,e_R) = \int\!\!\int_{C_A} [\int_L f(l|d,e_R) \cdot f_A(\underline{c}_A|a,l,e_R)] \cdot u_A(\underline{c}_A)\,d\underline{c}_A \tag{10}$$

To express all expected utilities as a function of r, the model postulates that the developer and impactee maximize the guesses of their expected utility given r and d. The maxima d(r) and a(d) are defined as

$$EU_D[d(r),r,e_R] = \max_D EU_D(d,r,e_R) \tag{11}$$

$$EU_A[a(d),d,e_R] = \max_A EU_A(a,d,e_R) \tag{12}$$

Finally, the model computes the total expected utility of r as a weighted average of the expected utility accruing to the regulator and the subsequent maximum utilities accruing to the developer and the impactee:

$$AEU(r,e_R) = \lambda_R EU_R(r,e_R) + \lambda_D EU_D[d(r),r,e_R) + \lambda_A EU_A\{a[d(r)],d(r),e_R\}. \tag{13}$$

The model defines the optimal regulation $r^* = (sl^*,sm^*,ss^*)$ by

$$AEU(r^*,e_R) = \max_R AEU\ (r,e_R). \tag{14}$$

Further analysis could include a sensitivity analysis with respect to the main model parameters, and a value of additional information analysis.

POSSIBLE USES OF THE MODEL

The decision theoretic model for standard setting consists of various elements: structural submodules (three unit decision making, goal trees and decision trees), estimation submodules (judgemental probability models and techniques), evaluation submodules (v. Neumann and Morgenstern utility functions), and an overall optimization model which is adapted to the three decision unit situation in standard setting. These submodules together constitute a decision aiding system which can help the regulator in standard setting decisions. Each of the submodules has its own usefulness for decision aiding, for guiding the information collection, aggregation, and evaluation tasks in standard setting.

The structural submodules provide the regulator with a framework for his analysis of the standard setting problem. The view of the decision problem as an interaction of three decision making units helps the regulator to use the decision tree and goal tree approaches effectively to determine in a formal way which alternatives, objectives and consequences are relevant for the regulation problem.

The estimation submodules give the regulator an opportunity to guide his information collection tasks, quantify the uncertainty of the experts which provide information, and to aggregate and communicate these quantified judgements. Furthermore, the Bayesian view of judgemental probability in the decision model allows him to integrate future information about technologies or effects of pollutants.

The evaluation submodules help the regulator to quantify intangibles, and to make trade-offs and evaluations explicit. Vague concepts of "priorities" or "desirability" are replaced by quantitative judgements of values and utilities through v. Neumann and Morgenstern functions.

The mathematical model that optimizes the regulatory decision presents the regulator with a tool to analyse a variety of questions interactively with the computer. For example, the regulator may want to perform a value of information analysis to determine if it is worth to fund a research study on the effects of a pollutant on human health.

Whether or not these possible uses can be realized depends on the answers to the two questions:

- Do the model structure and assumptions make sense?
- Can the functions f and u actually be constructed in a real standard setting situation?

These questions can best be answered through real world applications of the model. Within our research on standard setting at IIASA such an attempt is presently being made by applying the system to setting standards on chronic oil discharges from North Sea oil production platforms.

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A DECISION THEORETIC MODEL FOR STANDARD SETTING AND REGULATION*

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Technical Report No. 6 written for the Volkswagenwerk Foundation Project, "Procedures for the Establishment of Standards"

*Published also as a Research Memorandum at the International Institute for Applied Systems Analysis, Laxenburg, Austria, 1978.

ABSTRACT

This paper presents a decision theoretic model which was developed to aid regulatory agencies in standard setting and regulation tasks. The one stage three decision maker model encompasses the decision making of a regulator, a developer, and an impactee unit. Each decision unit is assumed to follow a basic decision model, which is a combination of a probability model, a difference value judgment model, and an expected utility model. The developer unit is linked to the regulator unit through possible detections of violations of a regulation and sanctions. The impactee unit is linked to the developer unit through pollution generating events stemming from the developer's actions, and the subsequent damages which may result from pollution.

This basic regulation model is then specified to safety and emission standard setting. Central in these specifications is a signal detection model which characterizes the uncertainty with which the regulator will detect or miss violations of his regulation. A multistage conditional probability model links the developer's actions, pollution generating events, amounts of pollutants, and possible effects on impactees.

A DECISION THEORETIC MODEL FOR STANDARD SETTING AND REGULATION

INTRODUCTION

Setting standards such as emission or ambient standards on chemicals polluting the air or the water is one of the most widely used regulatory tools to limit the negative effects of industrial activities on human health, safety, and psychological well being. Scientists and administrators in regulatory agencies who have to set such standards agree that the task is exceedingly difficult. There usually exists a vast amount of uncertainty about the effects of pollutants on human well being. Crucial trade-offs have to be made among multiple objectives which are often conflicting such as engineering, economic, political and environmental objectives. Conflicting interest groups are involved in standard setting each backing their case with different expert reports.

Since standard setting decisions are important, recurrent and exceedingly difficult, several attempts have been made to develop procedures along which a regulator and his experts can organize their information collection and evaluation tasks. Several authors have suggested and applied cost benefit analysis to solve the problem of standard setting and regulation (see, for example, Dorfman and Jacoby, 1970; North and Merkhofer, 1975; Karam and Morgan, 1975). But decision makers and scientists are often skeptical about the use of cost benefit analysis for such complex problems. The main reasons for this skepticism are that many values can not be expressed in Dollar terms and that the political character of the decision process is not taken into account (see, for example, Holden, 1966; Majone, 1976; Reports by the US Academy of Sciences, 1975; and by the National Research Council, 1977).

To aid regulators and scientists in standard setting tasks, therefore, new procedures and methods are called for. These procedures could include new institutional mechanisms (e.g. public participation, science courts, etc.) and new "softer" modeling approaches (e.g. decision theory and game theory). This paper concentrates on the second type of procedural innovation. It presents a formal decision theoretic model that was designed to help regulators to structure a standard setting task, to express uncertainties in a quantifiable form, and to evaluate alternative regulation and standards in the light of conflicting objectives. The paper is addressed mainly to decision theorists and operation researchers who are interested in the quantitative aspects of the model. Readers interested in the qualitative model structure are referred to von Winterfeldt (1978), and Fischer and von Winterfeldt (1978).

The paper is organized as follows. First the general decision theoretic model for a single decision maker will be developed. Readers familiar with measurement and decision theory on the level of DeGroot (1970), Fishburn (1970) and Krantz, Luce, Suppes and Tversky (1971) may wish to skip this part. The second section adapts the single decision maker model to a regulation model in which the decision making of a regulator, developer (producer) and impactee (sufferer) unit are linked. The third part of the paper details the general regulation model to the specific circumstances of standard setting.

BASIC DECISION MODEL FOR A SINGLE DECISION MAKER

The following mathematical formulation of the basic decision theoretic model is a modified version of the usual expected utility model (von Neumann and Morgenstern,1947) which is developed, for example in Raiffa and Schlaifer (1961), DeGroot (1970) and Fishburn (1970). It differs from the basic expected utility formulation in two aspects: first, it does not assume that conditional on event and action combinations final consequences can be predicted with certainty, but it leaves the possibility that there is a residual uncertainty about final consequences; second, it does not directly construct a von Neumann and Morgenstern utility function, but rather constructs it through an additive difference value function which can be shown to be functionally related to a von Neumann and Morgenstern utility function.

Let A be the set of decision alternatives (courses of action) with typical elements $a,b \in A$. Let S be a set of mutually exclusive and exhaustive events with typical elements $s,t \in S$. Let $C = \overset{n}{\underset{i=1}{X}} C_i$ be a subset of $\mathbb{R}^n$ which characterizes the possible consequences of act-event combinations. Typical elements of C are n-dimensional vectors $\underline{c}$, $\underline{d}$. Finally, let Z be the set of possible information sources with typical elements $y,z \in Z$.

The model assumes that the decision maker and his experts can quantify their uncertainty by a judgmental probability distribution (pd) over events and a probability density function (pdf) over consequences:

1) A probability distribution p which assigns to each event $s \in S$ a probability $p(s|a,z)$ depending on information source z and act a.

2) A probability density function f which assigns to each point $\underline{c} \in C$ a probability density $f(\underline{c}|a,s,z)$, depending on an act a, event s and information source z.

The residual uncertainty expressed in f can often be represented by independent marginal pdfs f_i. Therefore the following assumptions can be made:

$$f(\underline{c}|a,s,z) = \prod_{i=1}^{n} f_i(c_i|a,s,z) \quad , \tag{1}$$

where $c_i \in C_i$.

The total uncertainty about consequences given an act a and information source z can then be expressed by the following pdf g :

$$g(\underline{c}|a,z) = \sum_{s \in S} p(s|a,z) \prod_{i=1}^{n} f_i(c_i|a,s,z) \quad . \tag{2}$$

The set of all such probability density functions will be called F with typical elements $f, g \in F$,

The model assumes further that the decision maker has preferences among probability density functions, which can be characterized by the ordered set $\langle F, \gtrsim \rangle$. The interpretation of "$f \gtrsim g$" is that "f is preferred to or indifferent to g". The assumption which will be made in the following is that <u>$\langle F, \gtrsim \rangle$ is an expected utility structure</u>, i.e. that $\gtrsim$ obeys the <u>axioms of expected utility theory</u> (see von Neumann and Morgenstern, 1947; Savage, 1954; Fishburn, 1970). Therefore, there exists a function u: $C \rightarrow \mathbb{R}$ such that for all $f, g \in F$

$$f \gtrsim g$$

if and only if

$$\int_C f(\underline{c}) u(\underline{c}) d\underline{c} \geqslant \int_C g(\underline{c}) u(\underline{c}) d\underline{c} \quad . \tag{3}$$

Next, the model assumes that the decision maker or his experts can express their strength of preferences of one consequence over another, which can be characterized by the ordered set $\langle C \times C, \dot{\gtrsim} \rangle$. The interpretation of "$(\underline{c},\underline{d}) \dot{\gtrsim} (\underline{c}',\underline{d}')$" is that "the degree of preference of $\underline{c}$ over $\underline{d}$ is larger or equal to the degree of preference of $\underline{c}'$ over $\underline{d}'$". In the following the assumption will be made that <u>$\langle C \times C, \dot{\gtrsim} \rangle$ is an algebraic difference structure</u> i.e. that <u>$\dot{\gtrsim}$ obeys the axioms of algebraic difference measurement</u> (Suppes and Winet, 1955; Krantz, Luce, Suppes and Tversky, 1971). It follows that there exists a function v: $C \rightarrow \mathbb{R}$ such that for all $\underline{c},\underline{d},\underline{c}',\underline{d}' \in C$

$$(\underline{c},\underline{d}) \dot{\gtrsim} (\underline{c}',\underline{d}')$$

if and only if

$$v(\underline{c}) - v(\underline{d}) \geqslant v(\underline{c}') - v(\underline{d}') . \tag{4}$$

Both functions v and u express some evaluation aspect about consequences C, but there is no basis in the assumptions behind (3) and (4) that would establish a relationship between them. In particular there is no reason to assume that the expectation of v preserves the preference order of $f, g \in F$. To distinguish, u will be called a utility function, and v will be called a value function.

The remainder of the model description for the single decision maker will be concerned with establishing decomposition forms of u and v and functional relationships between u and v. In particular, independence assumptions on preferences among pdfs and on preference strength judgments will be made that lead to additive or multiplicative decompositions of u and v into single consequence functions u_i and v_i. Some further assumptions will then be used to show that the relationship between u and v must be either linear or exponential.

The reason for this type of model formulation is largely pragmatic. Assuming that v is additive, single consequence difference value functions v_i can be assessed through rather simple preference strength judgments. Given that u and v are related by a simple function, one can then construct a utility function u from v by assessing the parameters of the transformation either through sensitivity analysis or by asking a few simple insurance type questions. Thus the construction process, while ending with a von Neumann and Morgenstern utility function, circumvents the sometimes awkward lottery assessment methods which would have to be used otherwise. In addition, this type of model has the advantage of separating clearly between concepts of "marginal utility" (which has a place in the algebraic difference structure) and "risk attitude" (which has a place in the expected utility structure).

As a first step the preference relation and the preference strength relations are coupled. Preferences $\gtrsim$ are defined over F, but they can easily be applied to C by defining

$$\underline{c} \overset{'}{\gtrsim} \underline{d}$$

if and only if

$$f(\underline{c}) = g(\underline{d}) = 1 \text{ and } f \gtrsim g \quad . \tag{5}$$

Another induced preference relation can be defined by

$$\underline{c} \overset{''}{\gtrsim} \underline{d}$$

if and only if

$$(\underline{c},\underline{d}) \overset{*}{\gtrsim} (\underline{d},\underline{d}) \quad . \tag{6}$$

Nothing guarantees that $\overset{'}{\gtrsim} = \overset{''}{\gtrsim}$. However, from a judgmental point of view this equality seems plausible. Therefore, the following assumption will be made:

For all $\underline{c},\underline{d} \in C$

$$\underline{c} \overset{'}{\gtrsim} \underline{d}$$

if and only if

$$\underline{c} \overset{''}{\gtrsim} \underline{d} \quad . \tag{7}$$

If there are no ambiguities $\gtrsim$ will from now on be substituted for $\overset{'}{\gtrsim}$ and $\overset{''}{\gtrsim}$.

From definitions (5) and (6) and from assumption (7) it is obvious that there exists a functional relationship between u and v and that this relationship must be monotonically increasing:

$$\begin{array}{rll} u(\underline{c}) \geqslant u(\underline{d}) & \text{if and only if} & \\ & & \text{(by 3 and 5)} \\ \underline{c} \overset{'}{\gtrsim} \underline{d} & \text{if and only if} & \\ & & \text{(by 7)} \\ \underline{c} \overset{''}{\gtrsim} \underline{d} & \text{if and only if} & \\ & & \text{(by 6)} \\ (\underline{c},\underline{d}) \overset{\bullet}{\gtrsim} (\underline{d},\underline{d}) & \text{if and only if} & \\ & & \text{(by 4)} \\ v(\underline{c}) \geqslant v(\underline{d}), & \text{for all } \underline{c},\underline{d} \in C. & \end{array} \tag{8}$$

Thus u and v are both order preserving functions for preference over C and by the uniqueness of such functions (see Suppes and Zinnes, 1963; Krantz, Luce, Suppes, and Tversky, 1971) two such functions must be related through a monotonically increasing transformation u=h(v).

Next, the assumption will be made that v is additive:

$$v(\underline{c}) = \sum_{i=1}^{n} v_i(c_i), \quad c_i \in C_i \quad . \tag{9}$$

In the appendix an independence condition for $\overset{\bullet}{\gtrsim}$ is defined and a proof is given that this independence condition is sufficient for (9).

With respect to the decomposition of u, two possibilities are considered in the model:

Either

$$u(\underline{c}) = \sum_{i=1}^{n} u_i(c_i) \tag{10}$$

or

$$1+ku(\underline{c}) = \prod_{i=1}^{n} [1+ku_i(c_i)] \quad . \tag{11}$$

(10) and (11) are the classic decomposition forms of u in expected utility theory, and independence assumptions and proofs for these forms can be found, for example in Fishburn, 1970; Keeney, 1974; Keeney and Raiffa,1976.

The uniqueness property of additive value functions (see Fishburn, 1970; Krantz, Luce, Suppes and Tversky, 1971) states that any two additive order preserving functions defined on C must be related by a positive linear transformation. From this uniqueness property and assuming (9) and (10) the following relationship between v and u results:

$$u = \alpha v + \beta \quad \text{for some} \quad \alpha > 0,\ \beta \in \mathbb{R} \quad . \tag{12}$$

Again using the uniqueness property, but this time assuming (9) and (11) an exponential relationship results:

$$u = \frac{1}{k} e^{\alpha v+\beta} - \frac{1}{k} \quad \text{for some } \alpha > o, \beta \text{ if } k > o \;, \qquad \alpha < o, \beta \text{ if } k < o \quad . \tag{13}$$

In (13) α is a risk attitude parameter which, unlike usual risk parameters (see Raiffa, 1968), is not confounded with marginal value considerations. Marginal value considerations are expressed solely in v. The relationship between α and k is that $\alpha > o$ if $k > o$ and $\alpha < o$ if $k < o$. If $k = o$ then (12) holds. These results are proven and discussed in the appendix.

With definitions and assumptions (1) - (13) an evaluation function U can now be defined on AxZ which is consistent with a rational decision maker's preference and probability judgments:

$$U(a,z) = \int_C \left[\sum_{s \in S} p(s|a,z) \prod_{i=1}^{n} f_i(c_i|a,s,z)\right] h\left[\sum_{i=1}^{n} v_i(c_i)\right] d\underline{c}. \tag{14}$$

where h is either linear or exponential as defined in (12) and (13).

Given the appropriate choice of the functions like p, f, h, and v the decision rule that logically follows from the assumption made is:

select $a^* \in A$ with

$$U(a^*,z) = \max_{A} U(a,z). \tag{15}$$

Clearly one could also maximize over Z, the possible information sources. This would amount to a value of information analysis. Within the proposed model this could be done provided that the cost of information is included in the consequence space C. More specific models which assume additive costs of information and decisions could also be considered. Furthermore, additional information could be considered (e.g. a research study, an independent expert estimate) and a so called preposterior analysis of the value of additional information could be carried out (see Raiffa and Schlaifer, 1961).

The model allows a simple construction of the model functions. Decision theoretic techniques are available to construct pdfs, pds, and value functions in (14), and some of them have been developed to a high degree of sophistication. To construct pds odds comparison techniques can be used (see Spetzler and von Holstein, 1975). To construct pdfs over continuous random variables one would use either of two techniques: the fractile method (see Brown, Peterson and Kahr, 1974) or direct probability estimation techniques (see Spetzler and von Holstein, 1975). To construct the value function v rating and weighting techniques can be used which approximate quite closely the theoretically feasible techniques of using indifference judgments about value differences (Edwards, 1971). Within model (14) the transformation h from v into u is already so restricted that a few questions about insurance behavior and risk preferences should be sufficient to assess the parameters of the exponential form. If the transformation h is linear, one can use the value function directly as input into (14) without choosing transformation constants α and β, since utility functions are unique up to a linear positive transformation (see Krantz et al., 1971; Keeney and Raiffa, 1976).

THE DECISION MODEL FOR ENVIRONMENTAL REGULATION

In regulatory decision making the decision theoretic model described above becomes much more complicated, since several decision makers (or better: decision making units, organizations) are involved rather than an individual. These decision units have their own objectives, alternatives, and opinions. In a previous study on regulatory decision making (Fischer and von Winterfeldt, 1978) five main decision units were identified which enter into regulatory decision making. Generically they are called here:

1. the regulator unit
2. the developer unit

3. the impactee unit
4. the expert unit
5. the exogenous unit

The regulator unit consists of the people and institutions that have the responsibility for setting regulations as, for example, the Central Unit for Environmental Pollution in the UK, the State Pollution Control Agency in Norway, or the EPA in the US. The developer unit is defined as all those people and organizations whose span of decision alternatives is restricted by the regulation. In the impactee unit all those groups are combined whose activities or perceptions are impacted upon by the development activities. The expert unit consists of researchers and other experts that can provide information bearing on the regulatory problem. The exogenous unit is defined as those national or international organizations which constrain the decision making of the regulatory unit.

A good first approximation to the regulation problem was found to be achieved by considering the first three decision units only, and by identifying the elements of the expert unit with sources of information $z \in Z$. The resulting regulator-developer-impactee model has a structure which is schematically represented in Fig.1.

Each decision unit has its own alternatives (as well as sets of relevant events, information sources, consequence spaces, pdfs, value functions, etc.). If the regulator decides on a particular regulation alternative r, the decision making of the developer will be influenced by the possibility of sanctions if violations of the regulation r are detected. Thus the developer will respond to a regulation r by an action d(r) which may not be an action he would have taken without regulation.

Through pollution and their adverse effect on health and well being the developer influences the decision making of the impactee unit. In response to a developer decision d, the impactees will choose an action a(d) which may not be the action which they would have taken without the development activity.

The idea of the model is to determine optimal decisions d(r) and a[d(r)] for the developer and the impactee as a function of r, together with the associated utilities $U_R(r)$, $U_D[d(r)]$, $U_A\{a[d(r)]\}$. Further aggregation or Pareto optimality analysis may then be used to focus on a "good" value of r.

In the regulator-developer-impactee model each unit is represented by model (14). The notational specifications are given in Table 1.

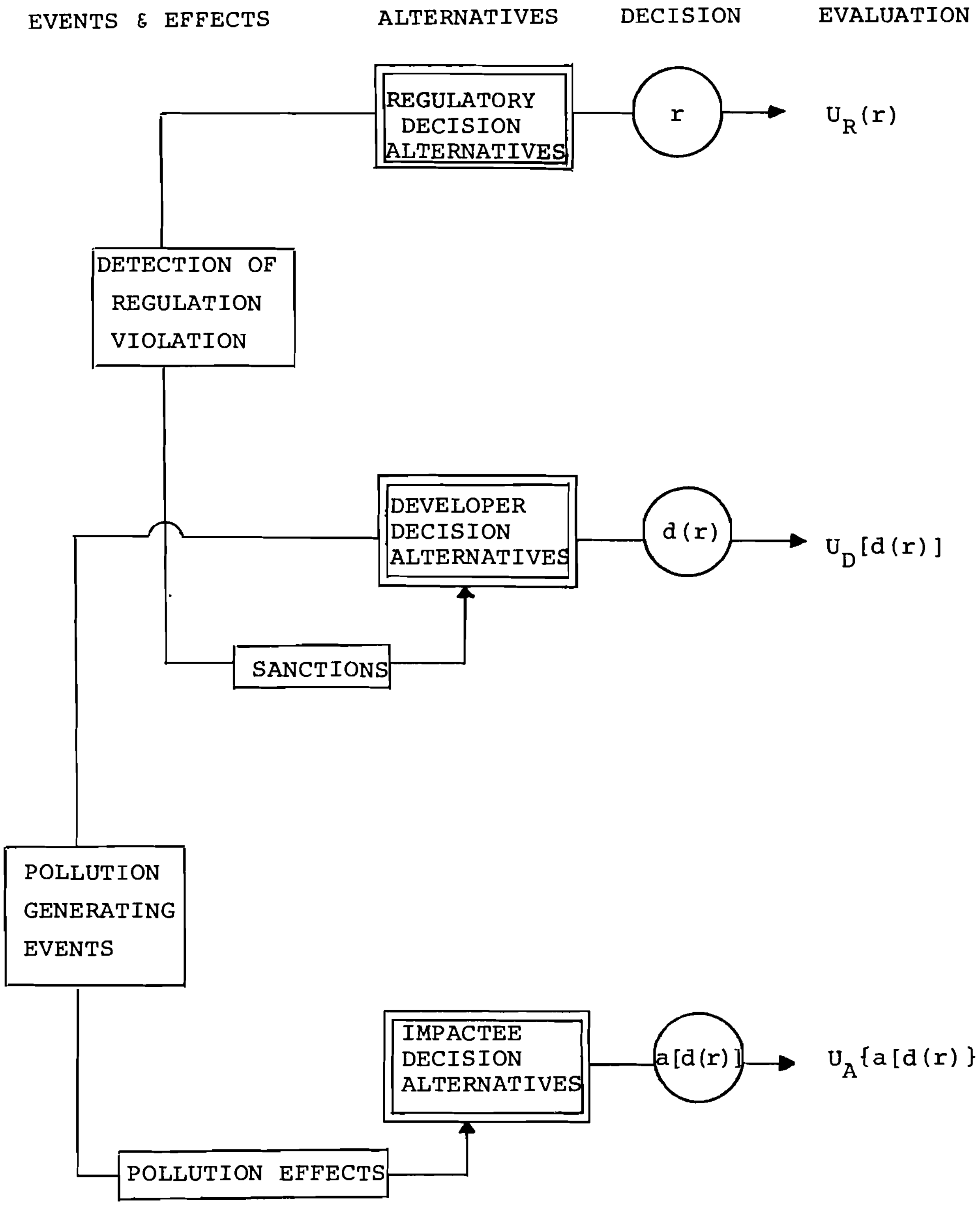

Figure 1. Schematic Representation of the Regulator-Developer-Impactee Model

Table 1. Notation for the Regulation Model

	Set	Element
Regulator's alternatives	R	r
Developer's alternatives	D	d
Impactee's alternatives	A	a
Regulator's consequences	C_R	$\underline{c}_R$
Developer's consequences	C_D	$\underline{c}_D$
Impactee's consequences	C_A	$\underline{c}_A$
Regulator's events	S_R	s_R
Developer's events	S_D	s_D
Impactee's events	S_A	s_A

The functions, pds, and pdfs $p_R, p_D, p_A, f_R, f_D, f_A, v_R, v_D, v_A$, h_R, h_D, h_A are also assumed. For simplicity the information sources Z_R, Z_D, Z_A are not spelled out any more and it is implicitly understood that all pds and pdfs of a decision unit are conditioned on an element $z_.$ from the information source $Z_.$ belonging to his unit.

Without specifying interlinkage models "detection of violation and sanction" and "pollution generation and effects" the evaluation of acts for each decision unit according to model (14) depends on the acts of other units, since $p_.$ and $f_.$ are dependent on all acts:

$$U_.(r,d,a) = \int_{C_.} [\sum_{s_. \in S_.} p_.(s_.|r,d,a) f_.(\underline{c}_.|r,d,a,s_.)] \cdot h_.[v_.(\underline{c}_.)] d\underline{c}_. \quad (15)$$

where "." stands for R, D, and A respectively. Let Q_o and Q_1 denote the events "non-detection" and "detection" of a regulation violation, and let $s_1, s_2, \ldots, s_K$ be the set of pollution generating events (e.g. explosions, normal operation of equipment, etc.). The model defines $S_R = \emptyset$, $S_D = \{Q_o, Q_1\}$ and $S_A = \{s_1, s_2, \ldots, s_K\}$. The following crucial independence assumptions are now made to simplify (15):

$$f_R(\underline{c}_R|r,d,a) = f_R(\underline{c}_R|r) \quad , \quad (16)$$

$$p_D(Q_o|r,d,a) = p_D(Q_o|r,d) = 1 - p_D(Q_1|r,d) \quad , \tag{17}$$

$$f_D(\underline{c}_D|r,d,a,Q_k) = f_D(\underline{c}_D|r,d,Q_k) \; k = 1,2 \quad , \tag{18}$$

$$p_A(s_j|r,d,a) = p_A(s_j|d) \quad j = 1,2,\ldots.,K \quad , \tag{19}$$

$$f_A(\underline{c}_A|r,d,a,s_j) = f_A(\underline{c}_A|d,a,s_j) \quad j = 1,2,\ldots.,K \; . \tag{20}$$

Verbally, these assumptions mean that:

(16) *The regulator's consequences only depend on his own action;*

(17) *Detection probabilities do not depend on the impactee's action;*

(18) *The developer's consequences do not depend on the impactee's action;*

(19) *Pollution generating event probabilities depend only on the developer's action;*

(20) *The impactee's consequences do not depend on the regulator's action.*

Therefore (15) can be written as follows:

$$U_R(r) = \int_{C_R} f_R(\underline{c}_R|r) \cdot h_R[v_R(\underline{c}_R)]\, d\underline{c}_R \quad , \tag{21}$$

$$U_D(d,r) = \int_{C_D} [\sum_{k=o}^{1} p_D(Q_k|r,d) f_D(\underline{c}_D|r,d,Q_k)] h_D[v_D(\underline{c}_D)]\, d\underline{c}_D , \tag{22}$$

$$U_A(a,d) = \int_{C_A} [\sum_{j=1}^{K} p_A(s_j|d) f_A(\underline{c}_A|d,a,s_j)] h_A[v_A(\underline{c}_A)]\, d\underline{c}_A \quad . \tag{23}$$

Independence of C_i, additivity of v, and the linear or exponential form of h lead to a further refinement of (21) - (23). Defining the optimal decisions of the developer and the impactee by

$$d(r): U_D[d(r),r] \geq U_D(d,r) \quad \text{for all } d \in D \quad , \tag{24}$$

$$a(d): U_A[a(d),d] \geq U_A(a,d) \quad \text{for all } a \in A \quad , \tag{25}$$

model (14) applied to all three decision units allows the determination of the optimal responses of the developer and impactee to a regulation r as well as the associated utilities $U_R(r)$, $U_D[d(r),r]$ and $U_A\{a[d(r)],d(r)\}$ for all three units as a function of r. A Pareto optimality analysis can now be performed on the $U_{\bullet}$, or, as a final step, one could postulate for some λ.

$$\bar{U}(r)=\lambda_R \cdot U_R(r) + \lambda_D U_D[d(r),r] + \lambda_A U_A\{a[d(r)],d(r)\} \quad , \tag{26}$$

with the optimal regulation defined as

$$r^* : \bar{U}(r^*) \geqslant \bar{U}(r) \quad \text{for all} \quad r \in R \quad . \tag{27}$$

THE DECISION MODEL FOR STANDARD SETTING

Model (15) - (27) will now be adapted to the specific circumstances arising in environmental standard setting as a particular type of regulation. The regulator's alternatives R will be further specified and the detection and consequence probabilities of the regulator and developer will be decomposed.

First, the regulatory alternatives R in standard setting are defined to consist of a set of standards SL, a set of monitoring and inspection procedures SM, and a set of possible sanctions for violations SS. Typical elements will be labelled *sl*, *sm*, and *ss* respectively. The developer's alternatives D are thought of as technological processes, equipments, and operations to reduce pollution risks and hazards. The impactee's alternatives are not further specified, and often treated as a dummy variable, i.e. the impactees are considered "sufferers".

Two classes of standards can be distinguished: safety standards are set by the regulator on D directly in order to reduce the probability of undesired events S_A; emission standards are set on amounts or rates of pollution in those cases in which for any choice of $d \in D$ the event "normal operation" can be taken for granted, but when such normal operations generate a constant flow of pollution.

Safety standards are considered first. In this case SL is defined as the set of all subsets of D. Consequently, if $sl \in SL$, then $sl \subseteq D$. D is considered to exist of mutually exclusive elements.

A violation of *sl* occurs if $d \notin sl$, otherwise the regulation is adhered to. The model assumes that the regulator cannot perfectly discriminate whether $d \in sl$ or $d \notin sl$, because of the limits of his monitoring and inspection procedure $sm \in SM$. Instead of d the regulator perceives $\hat{d} \in D$ with a probability distribution $p(\hat{d}|d,sm)$ which depends on d and *sm* only. A regulation violation is detected if $\hat{d} \notin sl$, not detected if $\hat{d} \in sl$. Detection probabilities (17) can now be specified:

$$p_D(Q_o|r,d) = \sum_{\hat{d} \in sl} p(\hat{d}|d,sm) \quad , \tag{28}$$

$$p_D(Q_1|r,d) = \sum_{\hat{d}\notin sl} p(\hat{d}|d,sm) \quad . \tag{29}$$

This formulation leaves the possibility open that the regulator detects a violation ($\hat{d}\notin sl$) when in fact no such violation occurred ($d\in sl$) and vice versa. In signal detection theory (Green and Swets, 1967) these cases are called "false alarm" and "miss".

The safety standards model assumes further that consequences accruing to the developer as expressed in (18) depend on regulatory action only through detections and sanctions:

$$f_D(\underline{c}_D|r,d,Q_k) = f_D(\underline{c}_D|ss,d,Q_k) \qquad k = 0,1 \quad . \tag{30}$$

The total consequence distribution is then

$$f_D(\underline{c}_D|r,d) = \sum_{d\in sl} p(\hat{d}|d,sm) \cdot f_D(\underline{c}_D|ss,d,Q_o) \; +$$

$$+ \sum_{\hat{d}\notin sl} p(\hat{d}|d,sm) \cdot f_D(\underline{c}_D|ss,d,Q_1) \quad . \tag{31}$$

Turning now to the consequence distribution of the impactee, let $l\in L\subseteq\mathcal{R}$ be an amount or rate of pollution which is considered a random variable with pdf $f(l|s_j)$.

The safety standards model assumes that, given l, the consequences accruing to the impactees do not depend any more on d and s_j:

$$f_A(\underline{c}_A|d,a,s_j,l) = f_A(\underline{c}_A|a,l) \tag{32}$$

Therefore, (2) can be written as

$$f_A(\underline{c}_A|d,a,s_j) = \int_L f(l|s_j)\, f_A(\underline{c}_A|a,l)\,dl \quad . \tag{33}$$

Event probabilities s_j are assumed to be independent of impactee actions $a\in A$:

$$p_A(s_j|d,a) = p_A(s_j|d) \quad . \tag{34}$$

Therefore the total consequence distribution for the impactee unit is:

$$f_A(\underline{c}_A|d,a) = \sum_{j=1}^{K} p_A(s_j|d)\int_L f(l|s_j) f_A(\underline{c}_A|a,l)\,dl\ . \tag{35}$$

These specifications leave the regulator unit unchanged, thus (21) still is the regulator's utility function. But substituting (31) into (22) and (35) into (23) gives new utility functions for the developer and the impactee. The three full utility functions for the safety standards model are

$$U_R(r) = \int_{C_R} f_R(\underline{c}_R|r)\ h_R[v_R(\underline{c}_R)]\,d\underline{c}_R\ , \tag{36}$$

$$U_D(r,d) = \int_{C_D} \{ [\sum_{\hat{d}\in sl} p(\hat{d}|d,sm)\ f_D(\underline{c}_D|ss,d,Q_o)\ + $$

$$+ \sum_{\hat{d}\notin sl} p(\hat{d}|d,sm)\ f_D(\underline{c}_D|ss,d,Q_1)]\}\cdot$$

$$\cdot\ h_D[v_D(\underline{c}_D)]\,d\underline{c}_D \tag{37}$$

$$U_A(d,a) = \int_{C_A} [\sum_{j=1}^{K} p_A(s_j|d)\ \int_L f(l|s_j)\ \cdot$$

$$\cdot\ f_A(\underline{c}_A|a,l)\,dl]\ h_A[v_A(\underline{c}_A)]\,d\underline{c}_A\ . \tag{38}$$

$d(r)$, $a[d(r)]$, and $\bar{U}(r)$ are determined as in (24), (25) and (26).

The emission standard version of the model defines sl not as a subset of D, but rather as an element of L. $sl\in L$ is interpreted as a maximum admissible amount of emission. S_A is now assumed to consist only of one element, say s_1, the event of "normal operation" of $d\in D$ for all d. This leads to

$$\begin{aligned} p_A(s_j|d) &= 1 \text{ for } j = 1 \\ p_A(s_j|d) &= 0 \text{ for } j \neq 1 \end{aligned} \tag{39}$$

Let $l\in L\subset\mathbb{R}$ be the amount or rate of emissions under s_1 and d with pdf $f(l|s_1,d)$. Since s_1 is the same for all developer decisions d, it will from now on be dropped.

The regulator tries to establish whether $l>sl$ (violation) or $l\leqslant sl$ (no violation). However, his monitoring and inspection procedure does not provide him with perfectly reliable information. Let $\hat{l}$ be a reading of emissions that the monitoring and inspection procedure registers. Let $g(\hat{l}|l,sm)$ be the reliability pdf that characterizes the quality of sm. Let q_o be the state "violation occurs $(l>sl)$" and q_1 the state "no violation occurs ($l \leqslant sl$)." The probabilistic relationship between the violation and detection states Q_o, Q_1, q_o, q_1 can be expressed in a fourfold table:

	q_o	q_1	
Q_o	p_{oo}	p_{o1}	$p_D(Q_o)$
Q_1	p_{1o}	p_{11}	$p_D(Q_1)$
	$p_D(q_o)$	$p_D(q_1)$	

Where the p_{ij}'s are the joint probabilities and the marginals are defined as:

$$p_D(q_o|d,sl) = \Pr[l \leqslant sl|d] \tag{40a}$$

$$p_D(q_1|d,sl) = \Pr[l > sl|d] \tag{40b}$$

$$p_D(Q_o|d,sl,sm) = \Pr[\hat{l} \leqslant sl|d,sm] \tag{40c}$$

$$p_D(Q_1|d,sl,sm) = \Pr[\hat{l} > sl|d,sm] \tag{40d}$$

From the above definitions the marginal probabilities (40a) and (40b) can be directly inferred:

$$p_D(q_o|d,sl) = \int_{l \leqslant sl} f(l|d)\,dl \quad , \tag{41a}$$

$$p_D(q_1|d,sl) = \int_{l > sl} f(l|d)\,dl \quad . \tag{41b}$$

The consequences that accrue to the developer depend, however, not on the true states of violation q_k but rather on the states of detection Q_k. These probabilities can usually not be inferred directly. But with the knowledge of $g(\hat{l}|l,sm)$ the joint probabilities p_{ij} can be determined as follows:

$$p_{oo} = \int_{\hat{l} \leqslant sl} \int_{l \leqslant sl} f(l|d) \cdot g(\hat{l}|l,sm)\, dl\, d\hat{l} \quad , \tag{42a}$$

$$p_{1o} = \int_{\hat{l} > sl} \int_{l \leqslant sl} f(l|d) \cdot g(\hat{l}|l,sm)\, dl\, d\hat{l} \quad , \tag{42b}$$

$$p_{o1} = \int_{\hat{l} \leqslant sl} \int_{l > sl} f(l|d) \cdot g(\hat{l}|l,sm)\, dl\, d\hat{l} \quad , \tag{42c}$$

$$p_{11} = \int_{\hat{l} > sl} \int_{l > sl} f(l|d) \cdot g(\hat{l}|l,sm)\, dl\, d\hat{l} \quad . \tag{42d}$$

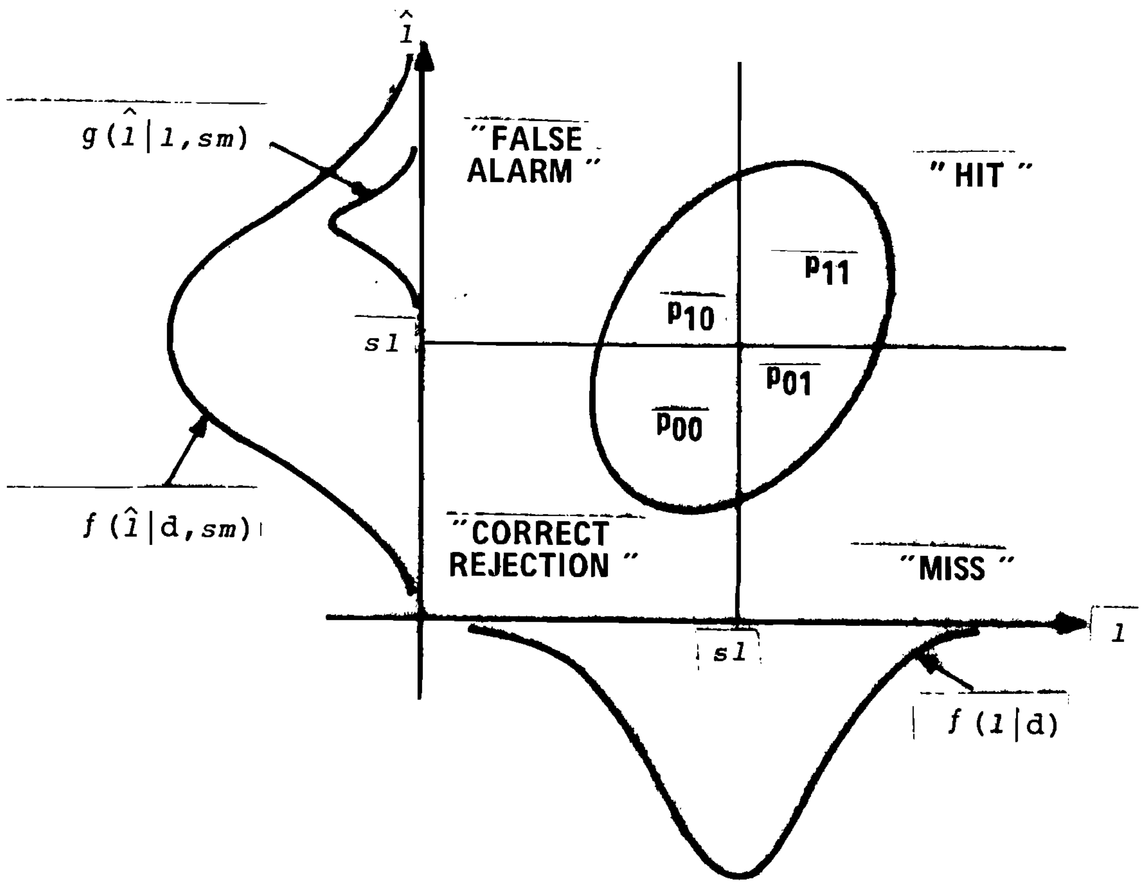

Figure 2. Graphic Representation of the Probabilistic Relationship between Violations and Detections (39-42)

The marginal probabilities of interest, $p_D(Q_o)$ and $p_D(Q_1)$, can now be determined as

$$p_D(Q_o|d,sl,sm) = \int_{\hat{l}\leqslant sl} \int_L f(l|d) \cdot g(\hat{l}|l,sm)\, dl d\hat{l} \quad , \quad (43a)$$

$$p_D(Q_1|d,sl,sm) = \int_{\hat{l}>sl} \int_L f(l|d) \cdot g(\hat{l}|l,sm)\, dl d\hat{l} \quad . \quad (43b)$$

These detection (non-detection) probabilities refer to a fixed time interval Δt. One could assume that this time interval is the lifetime of the plant. However, it is more realistic to assume that Δt is some "normal" time interval of inspection (e.g. one week). Then, over the lifetime of the plant T there will be $T/\Delta t$ possibilities for such detection. Let $n \cong T/\Delta t$; assuming that detection probabilities are constant over time and independent, the probability of no detection during T, labelled Q_o^n, is

$$p_D(Q_o^n|d,sl,sm) = p_D(Q_o|d,sl,sm)^n \quad . \tag{44a}$$

The probability of at least one detection during T, labelled Q_1^n, is

$$p_D(Q_1^n|d,sl,sm) = 1 - p_D(Q_o^n) \quad . \tag{44b}$$

The consequences accruing to the developer during T as expressed in f_D are now assumed to depend only on d,ss, and Q_k^n. (That is, as far as the developer is concerned, one detection is as bad as n):

$$\begin{aligned} f_D(\underline{c}_D|d,r) &= p_D(Q_o^n|d,sl,sm) \cdot f_D(\underline{c}_D|ss,d,Q_o^n) \;+ \\ &+ p_D(Q_1^n|d,sl,sm) \cdot f_D(\underline{c}_D|ss,d,Q_o^n) \;= \\ &= [\int_{\hat{l}\leqslant sl} \int_L f(l|d) \cdot g(\hat{l}|l,sm)\, dl d\hat{l}]^n \cdot \\ &\cdot f_D(\underline{c}_D|ss,d,Q_o^n) \;+ \\ &+\{1-[\int_{\hat{l}\leqslant sl} \int_L f(l|d) \cdot g(\hat{l}|l,sm)\, dl d\hat{l}]^n\} \cdot \\ &\cdot f_D(\underline{c}_D|ss,d,Q_1^n) \quad . \end{aligned} \tag{45}$$

As a further decomposition of the impactee's pdf f_A is concerned, the emission standards model follows the route of the safety standards model as expressed in (35). However, in emission standards, the picture gets simplified since $p_A(s_1|d) = 1$. Therefore, (35) becomes

$$f_A(\underline{c}_A|d,a) = \int_L f(l|d) \cdot f_A(\underline{c}_A|a,l)\, dl \quad . \tag{46}$$

Equation (46) expresses the usual view of the pollution problem as mediated through the levels of discharges into the environment. For each specific level probable consequences follow for the impactees. Equation (46) is the first step in the direction of decomposing the chain of events from emissions, over ambient distributions to final value relevant consequences. One could include another probability density function over ambient levels. One could then condition the consequence distribution f_A on ambient levels, rather than on emission levels, and define the total consequence distribution as a mixture of emission, ambient and the respectively conditioned consequence distribution for given ambient levels.

The full utility equations can now be written out for the emission standards model:

$$U_R(r) = \int_{C_R} f_R(\underline{c}_R|r)\ h_r[v_R(\underline{c}_R)]\ d\underline{c}_R \tag{47}$$

$$\begin{aligned} U_D(r,d) = \int_{C_D} & \{p_D(Q_o|d,sl,sm)^n\ f_D(\underline{c}_D|ss,d,Q_o^n)\ + \\ & + [1-p_D(Q_o|d,sl,sm)^n] f_D(\underline{c}_D|ss,d,Q_1^n)\} \cdot \\ & \cdot h_D[v_D(\underline{c}_D)]\ d\underline{c}_D \ . \end{aligned} \tag{48}$$

$$\begin{aligned} U_A(d,a) = \int_{C_A} & [\int_L f(l|d) \cdot f_A(\underline{c}_A|a,l)\ dl] \cdot \\ & \cdot h_A[v_A(\underline{c}_A)]\ d\underline{c}_A \ . \end{aligned} \tag{49}$$

In summary, regulation alternatives were decomposed into standards, monitoring and inspection procedures, and sanctions. Safety standards were considered to be set directly on D. The monitoring and inspection procedure was assumed to be fallible leading to the possibilities of misses or false alarms in regulation violation detection. This was the main vehicle for decomposing f_D. To decompose f_A it was assumed that pollution amounts are fully determined by s_j, the pollution generating events, and the developer decision d. Pollution amounts and impactee action alone determine the probable consequences for the impactee. The final utility equation for the safety standards model are (36-38).

The emission standard model, (47) - (49) assumed that a maximum level of emissions *sl* is set by the regulator, that pollution generating events S_A are reduced to "normal operation" for each d, that actual levels produced by d are probabilistic and can only be detected with error. This led to a rather specific definition of detection probabilities and a decomposition of f_D. f_A was decomposed as in the safety standard model but it was simplified, since pollution generating events were reduced to normal operations.

POSSIBLE USES OF THE DECISION THEORETIC MODEL

The model that has been developed here is a decision theoretic formulation of the regulatory decision problem in standard setting and it gives a stepwise solution of how a regulator might think of going through the standard setting task to solve the problem. The model prescribes an "optimal" standard setting solution that is consistent with the decision maker's preferences and opinions. However, the question arises of how the model can be applied in its full complexity to a real problem, and what its main uses and limitations are. There seem to be five ways in which a regulator could benefit from the use of the model. The model can help

- to structure the regulation problem;
- to enable regulators and their experts to express uncertainties, and intangibles in quantitative forms;
- to make trade-offs explicit;
- to identify a set of good regulatory solutions;
- to allow a study of the sensitivities of the regulatory solution to conflicting opinions, values, and information.

Structuring the Regulation Problem

The model provides a cognitive structure or roadmap along which the regulator can organize his thinking. Some of the main distinctions that the model makes are those between consequences, their values and probabilities, and between three decision-making units involved in regulation. Even if none of the further steps could be achieved (quantitative estimation of probability density functions, quantification of values) the model could already in this respect be an aid for the regulator.

Enabling Regulators and Experts to Express Uncertainties and Intangibles in a Quantitative Form.

In this presentation of the model the details of the actual quantification procedures were not given, although some methods were outlined on p.8. But through its simplifying assumptions the model is designed such that regulators and their experts can, in principle, perform the actual quantification tasks, both on the uncertainty and the value side. The model requires at no step to construct functions that would be very difficult to assess in practice (although by doing so it had to make some severe simplifying assumptions). The tools for such quantification exist and have been extensively explored in the laboratory and in real world decision problems. It remains a task of a model application to see how far one can go with the actual quantification steps in standard setting and regulations.

Making Trade-Offs Explicit

The model requires this directly by postulating quantitative trade-offs within each decision making unit (the v_i in the additive value function $v_.$), by postulating a parametrized risk transformations (the $h_.$), and by postulating trade-offs among decision units (the λ_i). Again, the question remains whether these weights and parameters that reflect the trade-offs can be realistically assessed. But the literature on multi-attribute utility theory and its applications indicate that a quantification of such trade-offs can be feasible.

Identifying a set of good standard solutions

The general regulation model (21-23), the safety standards model (36-38), and the emission standards model (47-49) all end up with the three utilities accruing to the regulator, the developer, and the impactee respectively, given that the latter two units take actions which are "optimal" in the decision theoretic sense. The search for a good standard solution can begin with an examination of the optimal developer and impactee responses $d(r)$ and $a[d(r)]$. Then a Pareto optimality analysis can be applied on U_R, U_D, and U_A to eliminate obviously unsatisfactory standards. Finally a weighting scheme such as (26) can be used to further explore the changes in utility as a function of r.

Allowing a Study of the Sensitivities of the Regulatory Solution to Conflicting Values, Opinions, and Degrees of Information

In each step the values, trade-offs, and probability density functions were made explicit based on a set of information that the regulator had at hand. Each of these parameters and information variables can be pushed around to see in which areas the model is most sensitive. For example, one could run the whole model based on some information provided by developers or impactees, to see if such different information sources would lead to different standard solutions. Furthermore, one could analyze if different weights or trade-offs would change the solution, etc. Finally, as an important output of the model one can compute the value of perfect information (see DeGroot, 1970), and the value of sample information. This could then be an important input for future budgeting decisions to set up research programs to improve standards.

Decision makers and analysts will probably find many formal and substantive limitations of the model presented here, ranging from criticisms that it is an overformalization of a very complex political process to specific criticisms of independence assumptions made in the model. Probably the most persuasive way to meet these criticisms are successful applications of the model. Within IIASA's research on standard setting one such application (on chronic oil discharge standards) has been completed, another one (on noise standards) is in process. In conjunction with these applications a final evaluation of the model presented here will be possible.

APPENDIX

In this appendix the additive decomposition of an algebraic difference function and the linear or exponential relationship between a von Neumann and Morgenstern function u and the difference function v will be derived from behavioral axioms.

Let $\langle C \times C, \overset{\circ}{\gtrsim} \rangle$ be an algebraic difference structure (Krantz et al., 1971). Let $\underline{c} = (c_1, c_2, \ldots, c_i, \ldots, c_n) \in C$ and let $(\underline{c}, \underline{d}) \overset{\circ}{\sim}$ $(\underline{c}', \underline{d}')$ be defined as $(\underline{c}, \underline{d}) \overset{\circ}{\gtrsim} (\underline{c}', \underline{d}')$ and $(\underline{c}', \underline{d}') \overset{\circ}{\gtrsim} (\underline{c}, \underline{d})$.
C_i is said to be <u>difference value independent</u> of $\underset{j \neq i}{X} C_j$ if and only if for all $c_i, d_i \in C_i, c_j^o, d_j^o \in C_j$, $j \neq i$ the following condition holds:

$$
\begin{aligned}
&[(c_1^o, c_2^o, \ldots, c_{i-1}^o, c_i, c_{i+1}^o, \ldots, c_n^o) \quad , \\
&\ (c_1^o, c_2^o, \ldots, c_{i-1}^o, d_i, c_{i+1}^o, \ldots, c_n^o)] \\
&\qquad\qquad\qquad \overset{\circ}{\sim} \\
&[(d_1^o, d_2^o, \ldots, d_{i-1}^o, c_i, d_{i+1}^o, \ldots, d_n^o) \quad , \\
&\ (d_1^o, d_2^o, \ldots, d_{i-1}^o, d_i, d_{i+1}, \ldots, d_n^o)] \ .
\end{aligned}
\tag{50}
$$

<u>Decomposition of v</u>

Let $\langle C \times C, \overset{\circ}{\gtrsim} \rangle$ be an algebraic difference structure with difference value function v as defined in (4). If for all i=1, 2,...,n-1, C_i is difference value independent of $\underset{j \neq i}{X} C_j$, then there exist functions $v_i : C_i \to \mathbb{R}$ such that

$$v(\underline{c}) = \sum_{i=1}^{n} v_i(c_i) \quad . \tag{51}$$

It is clear that difference value independence is necessary for (51) to hold. To prove sufficiency an approach by Fishburn (1970) for additive utility functions is followed. Consider the following n-1 indifference equations which are a result of the difference value independence condition for arbitrary $c_i, c_i^o, \in C_i$

$$[(c_1,\ c_2^o,\ c_3^o,\ \ldots,\ c_i^o,\ \ldots c_n^o)\ ,$$
$$(c_1^o,\ c_2^o,\ c_3^o,\ \ldots,\ c_i^o,\ \ldots c_n^o)]$$
$$\succsim$$
$$[(c_1,\ c_2,\ c_3,\ \ldots,\ c_i,\ \ldots c_n)\ ,$$
$$(c_1^o,\ c_2,\ c_3,\ \ldots,\ c_i,\ \ldots c_n)]\ ; \qquad (52.1)$$

$$[(c_1^o,\ c_2,\ c_3^o,\ \ldots,\ c_i^o,\ \ldots c_n^o)\ ,$$
$$(c_1^o,\ c_2^o,\ c_3^o,\ \ldots,\ c_i^o,\ \ldots c_n^o)]$$
$$\succsim$$
$$[(c_1^o,\ c_2,\ c_3,\ \ldots,\ c_i,\ \ldots c_n)\ ,$$
$$(c_1^o,\ c_2^o,\ c_3,\ \ldots,\ c_i,\ \ldots c_n)]\ ; \qquad (52.2)$$

$$\vdots$$

$$[(c_1^o,\ c_2^o,\ \ldots,\ c_{i-1}^o,\ c_i,\ c_{i+1}^o,\ \ldots,\ c_n^o)\ ,$$
$$(c_1^o,\ c_2^o,\ \ldots,\ c_{i-1}^o,\ c_i^o,\ c_{i+1}^o,\ \ldots,\ c_n^o)]$$
$$\succsim$$
$$[(c_1^o,\ c_2^o,\ \ldots,\ c_{i-1}^o,\ c_i,\ c_{i+1},\ \ldots,\ c_n)\ ,$$
$$(c_1^o,\ c_2^o,\ \ldots,\ c_{i-1}^o,\ c_i^o,\ c_{i+1},\ \ldots,\ c_n)] \qquad (52.i)$$

$$\vdots$$

$$[(c_1^o,\ c_2^o,\ \ldots,\ c_i^o,\ \ldots,\ c_{n-2}^o,\ c_{n-1},\ c_n^o)\ ,$$
$$(c_1^o,\ c_2^o,\ \ldots,\ c_i^o,\ \ldots,\ c_{n-2}^o,\ c_{n-1}^o,\ c_n^o)]$$
$$\succsim$$
$$[(c_1^o,\ c_2^o,\ \ldots,\ c_i^o,\ \ldots,\ c_{n-2}^o,\ c_{n-1},\ c_n)\ ,$$
$$(c_1^o,\ c_2^o,\ \ldots,\ c_i^o,\ \ldots,\ c_{n-2}^o,\ c_{n-1}^o,\ c_n)]\ . \qquad (52.n-1)$$

Since $\langle C \times C, \overset{\circ}{\gtrsim} \rangle$ is an algebraic difference structure from (4) and (52) the following n-1 difference equations can be derived:

$$v(c_1,c_2^o,\ldots,c_i^o,\ldots,c_n^o) - v(c_1^o,c_2^o,\ldots,c_i^o,\ldots,c_n^o) =$$

$$= v(c_1,c_2,\ldots,c_i,\ldots,c_n) - v(c_1^o,c_2,\ldots,c_i,\ldots,c_n) \quad , \qquad (53.1)$$

$$v(c_1^o,c_2,c_3^o,\ldots,c_i^o,\ldots,c_n^o) - v(c_1^o,c_2^o,\ldots,c_i^o,\ldots,c_n^o) =$$

$$= v(c_1^o,c_2,c_3,\ldots,c_i,\ldots,c_n) - v(c_1^o,c_2^o,c_3,\ldots,c_i,\ldots,c_n), \quad (53.2)$$

$$\vdots$$

$$v(c_1^o,c_2^o,\ldots,c_{i-1}^o,c_i,c_{i+1}^o,\ldots,c_n^o) -$$

$$v(c_1^o,c_2^o,\ldots,c_{i-1}^o,c_i^o,c_{i+1}^o,\ldots,c_n^o) =$$

$$= v(c_1^o,c_2^o,\ldots,c_{i-1}^o,c_i,c_{i+1},\ldots,c_n) -$$

$$v(c_1^o,c_2^o,\ldots,c_{i-1}^o,c_i^o,c_{i+1},\ldots,c_n) \quad , \qquad (53.i)$$

$$\vdots$$

$$v(c_1^o,c_2^o,\ldots,c_i^o,\ldots,c_{n-2}^o,c_{n-1},c_n^o) -$$

$$v(c_1^o,c_2^o,\ldots,c_i^o,\ldots,c_{n-2}^o,c_{n-1}^o,c_n^o) =$$

$$= v(c_1^o,c_2^o,\ldots,c_i^o,\ldots,c_{n-2}^o,c_{n-1},c_n) -$$

$$v(c_1^o,c_2^o,\ldots,c_i^o,\ldots,c_{n-2}^o,c_{n-1}^o,c_n) \quad . \qquad (53.n-1)$$

Adding up the left and right terms and cancelling the fourth term of equation (53.i) against the third term of (53.i+1) for all i results in

$$\sum_{i=1}^{n-1} v(c_1^o,c_2^o,\ldots,c_{i-1}^o,c_i,c_{i+1}^o,\ldots,c_n^o) -$$

$$(n-1) \cdot v(c_1^o,c_2^o,\ldots,c_i^o,\ldots,c_n^o) =$$

$$= v(c_1,c_2,\ldots,c_i,\ldots,c_n) - v(c_1^o,c_2^o,\ldots,c_i^o,\ldots,c_{n-1}^o,c_n) \quad (54)$$

Defining $\underline{c}^o = (c_1^o, c_2^o, \ldots, c_i^o, \ldots, c_n^o)$ and

$$v_i(c_i) = v(c_1^o, c_2^o, \ldots, c_{i-1}^o, c_i, c_{i+1}^o, \ldots, c_n^o) - v(\underline{c}^o) \cdot \frac{(n-1)}{n}$$

gives the desired result:

$$v(c_1, c_2, \ldots, c_i, \ldots, c_n) = \sum_{i=1}^{n} v_i(c_i) \tag{55}$$

The next decomposition is a well known result in expected utility theory. It is based on the definition of utility independence introduced by Keeney (see Keeney, 1974; Keeney and Raiffa, 1976). In the present terminology utility independence can be formulated as follows.

Let $\langle F, \gtrsim \rangle$ be an expected utility structure as defined in (3). Let f^i, g^i be any two probability density functions with identical degenerate marginal probability densities $f_i(c_i) = g_i(c_i) = 1$ for some $c_i \in C_i$, but otherwise unrestricted. $\mathop{X}_{j \neq i} C_j$ is said to be utility independent of C_i if the preference among f^i and g^i does not depend on the specific value of c_i with unity density. Weaker formulations of utility independence can be found in Keeney (1974); Fishburn and Keeney (1974); von Winterfeldt and Fischer (1975); Keeney and Raiffa (1976). The result of utility independence together with the assumption of a von Neumann and Morgenstern utility function over C is the following decomposition:

Decomposition of u

Let u be a von Neumann and Morgenstern utility function over C. Let $\mathop{X}_{j \neq i} C_j$ be utility independent of C_i for all i. Then there exist functions $u_i: C_i \to \mathbb{R}$ such that either

$$u(\underline{c}) = \sum_{i=1}^{n} u_i(c_i) \tag{56}$$

or

$$1 + ku(\underline{c}) = \prod_{i=1}^{n} [1 + ku_i(c_i)] \quad . \tag{57}$$

The proof can be found in the papers cited above.

To relate u and v the following uniqueness theorem for additive order preserving functions is observed (see Krantz et al, 1971)*: If a function $v: C \to \mathbb{R}$ is additive and preserves

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*A similar proof based on constant risk aversion arguments has been provided by Pratt and Meyer in Keeney and Raiffa (1976).

the order of C, and if there is another function $v':C\to\mathbb{R}$ with the same property then there exist real numbers $\alpha,\beta,\alpha>0$ such that

$$v = \alpha v' + \beta .$$

This property, now used to relate u and v with respect to the orders $\overset{'}{\gtrsim}$ and $\overset{''}{\gtrsim}$ over C, which by (7) are identical.

If (51) and (56) hold u and v are both additive and order preserving functions from $C\to\mathbb{R}$.

Therefore, there exist $\alpha>0,\beta$ such that

$$u = \alpha v + \beta \quad . \tag{58}$$

If (51) and (57) hold, then first (57) needs to be transformed into an additive order preserving function before the uniqueness theorem can be applied. Two cases have to be distinguished: if $k>0$, then $1+ku$ in (57) is an order preserving function. If $k<0$ then $1+ku$ has an inverse order.

Consider $k>0$ first:

$$\ell n[1+ku] = \ell n[\prod_{i=1}^{n}(1+ku_i)] \quad , \tag{59}$$

$$\ell n[1+ku] = \sum_{i=1}^{n} \ell n(1+ku_i) \quad . \tag{60}$$

Since $(1+ku)$ is order preserving and ℓn is strictly increasing, $\ell n(1+ku)$ is also order preserving. Therefore (60) is an additive order preserving function. By uniqueness, there exist $\alpha>0,\beta$ such that

$$\begin{aligned} \ell n[1+ku] &= \alpha v + \beta \\ 1+ku &= e^{\alpha v+\beta} \end{aligned} \tag{61}$$

$$u = \frac{1}{k}e^{\alpha v+\beta} - \frac{1}{k} \tag{62}$$

where $k>0$ and $\alpha>0$.

If $k<0$ then $(1+ku)$ is an inverse order. Since ℓn is strictly increasing $\ell n(1+ku)$ is an inverse order, and

$$-\ell n[1+ku] = -\sum_{i=1}^{n} \ell n(1+ku_i) \tag{63}$$

is again an additive order preserving function. By uniqueness, there exist $\alpha>0,\beta$ such that

$$- \ell n[1+ku] = \alpha v + \beta \qquad (64)$$

$$1+ku = e^{-\alpha v-\beta}$$

$$u = \frac{1}{k} e^{-\alpha v-\beta} - \frac{1}{k} \qquad (65)$$

where $k<0$, $\alpha>0$.

(58), (62), and (65) are the desired results.

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A GAME-THEORETIC FRAMEWORK FOR DYNAMIC STANDARD SETTING PROCEDURES*

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ABSTRACT

This paper presents a game-theoretic approach to modeling environmental standard setting procedures under specific consideration of the dynamic conflict situation in environmental decisions. Three idealized decision units are considered, the regulator, producer and impactee units: The regulator has to fix the standard. This standard causes a financial burden to the producer, who releases pollutants to the environment. By means of the standard the impactee has to be protected against this pollution.

The starting point is a multistage model for a non-cooperative three person game. After the description of this model the range of its application is indicated by the cases of North-Sea oil, sulphur dioxide, carbon dioxide, and noise. Since any game-theoretic analysis includes the choice of a solution concept, a class of concepts is discussed. The last part of the paper contains a brief survey of the results of two multistage cases where the relevance of the solution concepts is demonstrated.

TABLE OF CONTENTS

A GAME-THEORETIC FRAMEWORK FOR DYNAMIC STANDARD SETTING PROCEDURES

INTRODUCTION

Since the end of the 1960s environmental agencies have been set up all over the world establishing guidelines and regulations that should help to limit effects of modern technologies that may be detrimental to the environment. New organizations, regulatory tools, standards, incentives, and procedures were rapidly introduced which often had a substantial impact on the industrial investment and operating costs as well as on the speed at which new technologies were introduced. After an initial period of zealous environmental decision making the time has come now to reflect on this development. Questions such as the following are raised both by environmental researchers and decision makers: How good are our procedures for assessing impacts on the environment? How well do we take uncertainties into account when making regulatory decisions? Are long-term environmental and economic effects of our decision making properly taken into account?

Researchers and experts of environmental agencies began to realize that the difficulties in environmental decision making often lead to decisions that are less rational than one would wish. The problem areas most often mentioned are the vast uncertainties that exist about the environmental effects of pollutants, the difficulty in assessing risks of accidents of scales never encountered before, the conflicting interests of groups involved in and affected by regulatory decision making, and the difficulty in assessing long-term environmental and economic effects. These problems call for new institutional and methodological approaches to environmental decision making (see National Academy of Sciences, 1975, National Research Council, 1977).

This paper presents a game-theoretic approach to the modeling of environmental standard setting decisions, considering specifically the dynamic conflict situation in environmental decisions.

Three decision-making units are considered in the game theoretic model: the regulator, producer, and impactee units; such a structure has in fact also been proposed in connection with risk analysis (H. Otway, P. Pahner, 1976). The *regulator*, who may consist of a regulatory agency where various administrative units and experts interact, has to fix a standard. This standard usually causes a financial burden to the *producer*, who may consist of several energy producers emitting gaseous pollutants, or any other enterprise polluting the environment. The standard serves to protect the *impactee* consisting of the population affected by the pollution.

Under special assumptions about the parties involved one arrives at a conflict among several people that belongs to the class of problems treated by game theory. The assumptions are essentially two: "Each individual has a utility-function that he strives to maximize;" and "Each individual is able to perceive the gaming situation." These two are often subsumed under the phrase "The theory assumes rational players" (R.D. Luce, H. Raiffa, 1957, ch. 1). The problem of how to arrive at utility functions from given preference patterns is dealt with by decision theory (see e.g. D. v. Winterfeldt, 1978, 1), and will not be discussed in this paper. Instead the purpose of this paper is to provide an appropriate game-theoretic framework for standard setting, and to discuss the value of the game-theoretic results for the problem.

The starting point is a multistage model for a game between the three players: regulator, producer, and impactee. It is hoped that the model is general enough to embrace some essential features of most problems of standard setting. Furthermore it should permit parameter analysis in a way that crucial uncertainties about health effects and economic development as well as about utility functions can be identified. This parameter analysis seems to be indispensable especially for the regulator's utility function, since his utility function should reflect both general economic considerations and detrimental effects of pollution on the population, the weights on both being highly arbitrary. Though essentially descriptive, these models should help the regulating authority structure the standard setting task, including such problems as whether and what research program to start, e.g. on health effects, in order to reduce crucial uncertainties. Furthermore they allow one to look at cases where technical or physical parameters dominate such that for all reasonable utility functions and existing uncertainties nearly the same results are obtained.

The models concentrate on long-term aspects or dynamic problems and rather neglect distribution and bargaining problems (see, e.g., Organization for Economic Co-operation and Development, 1976, and J.C. Harsanyi, 1977) although these can be included in principle.

The paper is organized as follows. First the model description is given. Then the range of applications is illustrated by cases such as North Sea oil, sulphur dioxide, carbon dioxide, and noise. The North Sea oil problem was treated as a detailed one-stage game model, and multistage models were developed for carbon dioxide and noise (D. v. Winterfeldt, 1978), (E. Höpfinger, D. v. Winterfeldt, 1978). The multistage cases are sketched thereafter.

Since there is a variety of different solution concepts for n-person games ($n \geq 2$), any game-theoretic analysis includes the choice of a solution concept. That is why a class of appropriate solution concepts are discussed: the equilibrium point for noncooperative games, Pareto-optimal points for essentially cooperative games, the "minimal distance from bliss-point" concept, and the Nash solution. Furthermore a hierarchical solution concept is given for cases where first the regulator announces his strategy and thereafter the producer. This two-level leadership concept may be regarded as normative.

At the end a brief survey is given of the results of the two multistage cases demonstrating the relevance of the solution concepts.

MODEL DESCRIPTION

The dynamic or multistage models developed below are *three-person games in extensive form.* The definition of such games is rather involved and, since the authors hope that the following description is sufficiently self-contained for a general definition of games in extensive form, they only refer to (J.C.C. McKinsey, 1952) and (G. Owen, 1968).

The Time-Discrete Game

It is assumed that only time periods or stages have to be considered instead of a time-continuum. Thus a game is played at each stage, and the player's strategies control not only the payoff but also the transition probabilities governing the game to be played at the next stage. Each component game is determined by the states of the play. For example s can contain the relevant physical state of the world, e.g., the amount of oil in the water, of sulphur dioxide in the air, and their distribution; or the relevant economic state. Other than with the more usual games where players make simultaneous and independent choices, *perfect information* is assumed for the component game by the following structure: At each stage the regulator makes his choice first, then the producer is informed about the regulator's choice and makes his choice, and finally the impactee learns about the other choices and makes his choice.

The play proceeds from component game to component game with the transition probabilities jointly controlled by the players. Since the transition probabilities are often not exactly known, subjective transition probabilities are admitted for the players which may differ from each other. The process of the play can be sketched as in Figure 1.

Let S denote the set of possible states. For each $s \in S$ the set of the regulator's choices or measures is denoted by $M_R(s)$. Let $M_P(s,m_R)$ denote the set of producer's measures or choices in the case of state s and the regulator's choice m_R. If the

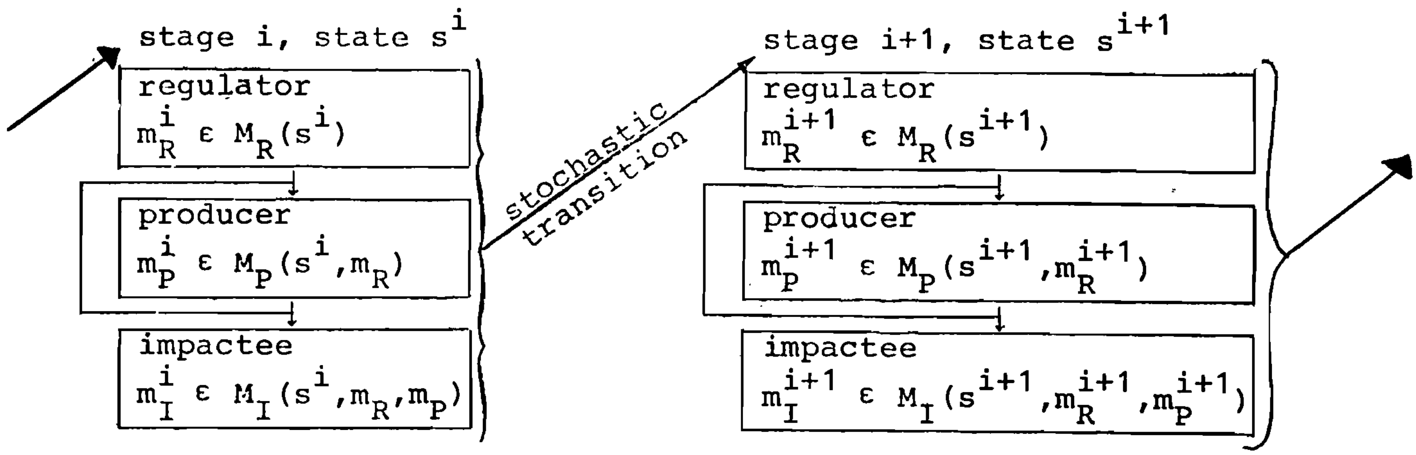


Figure 1. Transition from stage i to stage i+1.

producer chooses $m_P \in M_P(s,m_R)$, then $M_I(S^i,m_R,m_P)$ denotes the set of choices or measures possible for the impactee. Hence M_R is a map on S; M_P a map on $\{(s,m_R)|s\varepsilon S,m_R\varepsilon M_R(s)\}$; and M_I a map on $\{(s,m_R,m_P)|s\varepsilon S,m_R\varepsilon M_R(s),m_P\varepsilon M_P(s,m_R)\}$. Then $P_j(.|s,m_R,m_P,m_I)$ (j=R,P,I) denotes the subjective probability for the next state given state s and choices m_R,m_P,m_I. Strictly speaking $P_j(.|s,m_R,m_P,m_I)$ is a probability measure on the measurable space (S,∂), where ∂ is an appropriate σ-algebra that depends only on the last state and choices neglecting all previous states and choices. For each component game a utility function is given for each player:

$$U_j : \{(s,m_R,m_P,m_I)|s\varepsilon S,m_R\varepsilon M_R(s),m_P\varepsilon M_P(s,m_R),m_I\varepsilon M_I(s,m_R,m_P)\}\to R \quad ,$$

where $U_j(s,m_R,m_P,m_I)$ denotes the payoff to player j (j=R,P,I).

Games which may stop after finitely many stages can be included such that a permanent state is reached providing only one choice for each player and zero payoff for each. This is important in case one tries to approximate infinite stage games by finite stage games.

A play of the game is given by an infinite sequence $(s^1,m_R^1,m_P^1,m_I^1;\ s^2,m_R^2,m_P^2,m_I^2;\ \ldots)$ of states and decisions. Then one possibility for the payoff functions is given by

$$\underline{U}_j(\pi) := \sum_{i=1}^{\infty} \rho_j^i U_j(s^i, m_R^i, m_P^i, m_I^i) \qquad (j = R,P,I) \quad ,$$

where $0 < \rho_j \leq 1$ is a *discount factor* for player j. A second one is given by

$$\lim_{n\to\infty} \frac{1}{n} \sum_{i=1}^{n} U_j(s^i, m_R^i, m_P^i, m_I^i) \quad .$$

Since the latter suppresses the payoff of the first stages we shall only use the first. The discount factor ρ_j is the larger the more the future is regarded as important. In general, $\underline{U}_j(\pi)$ is well defined if $\rho_j < 1$. For special cases, however, $\underline{U}_j(\pi)$ is well defined for $\rho_j = 1$ because technical constraints such as limited resources of fuel limit the summation

$$\sum_i U_j(s^i, m_R^i, m_P^i, m_I^i) \quad .$$

In order to arrive at games that are not too complicated only stationary strategies have been considered. Thus a strategy σ_R of the regulator is a function of S providing always the same choice $\sigma_R(s) \varepsilon M_R(s)$ as soon as $s \varepsilon S$ occurs; a strategy σ_P of the producer is a function on $\{(s, m_R) \mid s \varepsilon S, m_R \varepsilon M_R(s)\}$ providing always the same choice $\sigma_P(s, m_R) \varepsilon M_P(s, m_R)$ as soon as (s, m_R) occurs; and analogously the impactee's strategy σ_I is a function on

$$\{(s, m_R, m_P) \mid s \varepsilon S, m_R \varepsilon M_R(s), m_P \varepsilon M_P(s, m_R)\} \quad ,$$

such that

$$\sigma_I(s, m_R m_P) \varepsilon M_I(s, m_R, m_P) \quad .$$

Given a strategy tuple $(\sigma_R, \sigma_P, \sigma_I)$, a subjective probability $P_j(\cdot \mid \sigma_R, \sigma_P, \sigma_I)$ over the space of possible plays is determined for each player. Under measurability conditions not specified each player can expect a payoff given by

$$V_j(\sigma_R, \sigma_P, \sigma_I) = \int \underline{U}_j(\pi) dP_j(\pi \mid \sigma_R, \sigma_P, \sigma_I) \qquad (j = R,P,I) \quad ,$$

where $\underline{U}_j(\pi)$ denotes the payoff in the case of play π.

Except for a solution concept and except for a mathematical discussion of the assumptions necessary for the well-behavior of the mathematical terms above, the model description is complete.

So far the population affected by pollution has been represented as a rational player with a utility function. This is no self-evident approach. Another possibility would be to represent the population by a *response function* based on its perception of the effects of pollution. But this can be done within the game-theoretic model given above in that the choice sets $M_I(s,m_R,m_P)$ contain one element only. If the impactee's payoff is not of interest one can drop the impactee and only consider the transition probabilities of regulator and producer. However, it is not easy in general to formulate a response function adequately describing the reactions of the population. One result of a three-person game-theoretic model may therefore consist in response functions that are special strategies of the impactee and are considered with some solution of the game.

Juridical procedures can be formalized within this framework at least by representing a court sentence as a transition from one state into another. Research programs on health effects and the impactee's attitude can reduce the range of M_I and make the transition law more exact, reducing, for example, the variance of a distribution relating to the transition.

Extensions

If the game has only finitely many stages and the sets of states and measures of all the players are finite, the game always has an equilibrium point in "pure" (nonstationary) strategies (see, for example, J. Rosenmüller, 1977), i.e. no random choices are necessary. This is due to the property of full information for all players. Nevertheless one may ask whether other orders of succession among the players' choices are appropriate. Firstly, this approach seems a suitable one since the regulator is often regarded as the most powerful player who usually is the first announcing his choices. Citizen groups usually only react to the regulator's or producer's decision. Secondly, an alternative order of succession can be included by introducing dummy choices and enlarging the state space by the players' last choices. Of course, this might yield a cumbersome model.

One arrives at much more complicated games if one considers strategies like "reduction by 20 percent of emission of a pollutant over five years" if there is no major change of economic or technical conditions. Due to a lack of time such a model has not been developed. Due to the stationary property of strategies, however, this model can increase the probability of emission reduction by 20 percent over five years thus reflecting a "mixed" strategy.

Bargaining of the players can be included (J.C. Harsanyi, 1977). Bargaining among the groups that are represented by the three players is not a major point of the game-theoretic model. Instead we rather start from the assumption that the groups have reached agreements. Thus, for example, an analysis like the one of (W. Richter, 1978) of the location of a public utility has not been carried over to detrimental facilities like nuclear plants using cooperative game theory where the players are the affected individuals. In the case of global pollution and local regulators, producers, and citizen groups, however, the local models are the basis for modeling the conflict situation among the groups of regulators.

RANGE OF APPLICATIONS

The following description of cases serves as an introduction into the variety of problems that can be treated within the framework outlined above.

North Sea Oil

Due to oil haulage in the North Sea there is now, even during normal operation, pollution by chronic oil discharges in addition to accidental oil spills.

Components of state: distribution of polluting oil in the North Sea, amount of oil raised in the previous year, amount of fish caught in the last previous year, recreation index of the coast, equipment and organization of the three players.

Choices:

a) Regulator: maximal amount of oil pollution, monitoring systems together with basic juridical measures (taxes), research programs on effects of pollution;

b) Producer: amount of oil to be raised during the next period, treatment, equipment, violation of standard;

c) Impactee: no action, aggression against oil company, changes of political leaders, fishermen drop their jobs, tourists avoid coasts.

Consequences, costs, and benefits: satisfication of standards of other nations, increase of gross national product, better balance-of-payments, decreased water quality, reduction of fishing and tourism.

Sulphur Dioxide

Regional pollution by burning fossil fuel.

Components of state: distribution of sulphur dioxide in the air, number of ills effected by sulphur dioxide, amount of sulphur dioxide produced in the previous year, distribution of population, attractivity factor of landscape, percentage of unemployed, gross national product,...

Choices:

a) Regulator: maximal amount of emitted SO_2 (including juridical basis), (taxes), monitoring, removal of producers, initiate research program on health effects, improvement of medical systems, help for migration of population, ...;

b) Energy producer: installation of filters, reduction of energy production, combustion of other fuels;

c) Impactee: migration, aggression against government or energy producer, civil action, vote to suspend government, reducing his own consumption of energy.

Consequences, costs, and benefits: employment, large gross national product, lung diseases, ultimately death.

Carbon Dioxide

Global pollution manifested as increased amount of carbon dioxide in the atmosphere.

Components of state: amount of atmospheric CO_2, temperature, high temperature catastrophe.

Choices:

a) Regulator: maximal amount of emitted CO_2 (including juridical basis);

b) Producer: amount of production of CO_2;

c) Impactee: aggression against energy producer or government, vote to suspend government, reduce energy consumption.

Consequences, costs, and benefits: employment, large gross national products, catastrophe.

Noise

A lot of industrial activities impose a noise problem on their environment. This description relates to the fast Shinkhansen train in Japan.

Components of state: maximum quantity of noise near the railway line, settlement in the vicinity of the railway line, layout of soundwalls, upper bound for speed of trains.

Choices:

a) Regulator: maximal quantity of speed or noise, order to build sound walls;

b) Producer (of noise): sound walls, reduced speed, dislocation of neighbors;

c) Impactee: complaints, petition to regulator, legal action against railway company.

Consequences, costs, and benefits: increased or decreased gross national product, dislocation of residents, health effects on residents.

EXAMPLES

The North Sea Oil problem as yet has only been treated as a detailed one-stage model by D. v. Winterfeld, (1978, 2). The study contains considerations that are difficult to handle within a genuine multistage model and is not discussed here further. It has turned out that the sulphur dioxide problem can only be treated adequately within a regional model including several pollutors, input-output analysis, and migration problems. Considering the lack of solutions and in the understanding of the basic structures of simpler cases, this problem has been postponed. In the short period of time available only studies on carbon dioxide and noise as dynamic games were carried out that are briefly outlined in this paper. Detailed descriptions can be found in (E. Höpfinger, 1978, 1) and (E. Höpfinger, D. v. Winterfeldt, 1978).

A Multistage Model for the Atmospheric Carbon Dioxide Problem

The effects of increased shares of carbon dioxide in the atmosphere are not well known. The conjectures that exist at present are rather contradictory. This model is based on the assumption that a continuous increase of CO_2 in the atmosphere beyond an unknown critical value, caused by the burning of fossil fuel, will lead to irreversible and large changes in the climate of the earth that are to be regarded as catastrophic. The regulator is assumed to be an international agency, and the group of all emitters of CO_2 as the producer.

The states of the game are

$\{(C,L)|C\geq 0,L\geq 0\} \cup \{k\geq 0\}$

where

C is the amount of carbon dioxide in the atmosphere;
L is the upper bound of emission of CO_2 during the period;
k is the critical value of the atmospheric CO_2-content.
Since the true critical value is unknown one has to consider the set of all possible critical values.

Let (C^1,L^1) denote the first state. The choices of the players in case of state (C,L) are the following:

The regulator chooses $0 \leq l \leq L$, with l denoting the upper bound of carbon dioxide emitted by the producer. The producer chooses $0 \leq a \leq l$, the amount of CO_2 to be emitted. The producer chooses the degree of pressure $0 \leq p \leq 1$ he wants to exert on the regulator. With probability pv the bound L is replaced by $\frac{L}{2}$, where $0 < v < 1$ is a fixed number.

For state k the choices of the players are $l = 0$, $a = 0$, $p = 0$.

By assumption the critical value is not known and further information is not available. Hence all three players may have different conjectures denoted by C_I, C_P, and C_R. For simplicity C_P denotes the maximal amount of carbon dioxide in the atmosphere if all fossil fuel is burnt.

Given state (C,L) and the choices (l,a,p) the following states are possible at the next stage:

$(C+\beta a,L)$, $\quad (C+\beta a,\frac{L}{2})$, $\quad \{k\geq C\}$,

with βa denoting that part of the carbon dioxide emitted remains in the air. β is assumed to be constant. The subjective probabilities P_R,P_P,P_I for the new states are:

New State	P_R	P_P	P_I
$(C+\beta a,L)$	0 if $C\leq C_R<C+\beta a$ 1-pv if $C+\beta a\leq C_R$	1-pv	0 if $C\leq C_I<C+\beta a$ 1-pv if $C+\beta a\leq C_I$
$(C+\beta a,\frac{L}{a})$	0 if $C\leq C_R<C+\beta a$ pv if $C+\beta a\leq C_R$	pv	0 if $C\leq C_I<C+\beta a$ pv if $C+\beta a\leq C_I$
C_R	1 if $C\leq C_R<C+\beta a$		
C_I			1 if $C\leq C_I<C+\beta a$

State k cannot be changed.

The transition from state s to state t has the utility

$U_j'(s;l,a,p,;t)$ for player j=R,P,I.

$$U_j'(C,L;l,a,p;C+\beta a,M) = c_1l+c_2a+c_3p \quad (M=L,\tfrac{L}{2})$$
$$U_R'(C,L;l,a,p;k) = c_1l+c_2\frac{k-C}{\beta}+c_3p+c_R$$
$$U_R'(k;o,o,o;k) = 0$$
$$U_P'(C,L;l,a,p;C+\beta a,M) = c_4a \quad (M=L,\tfrac{L}{2})$$
$$U_P'(C,L;l,a,p;k) = c_4\frac{k-C}{\beta} + c_P$$
$$U_P'(k;o,o,o;k) = 0$$
$$U_I'(C,L;l,a,p;C+\beta a,M) = c_5a+c_6p \quad (M=L,\tfrac{L}{2})$$
$$U_I'(C,L;l,a,p;k) = c_5\frac{k-C}{\beta} +c_6p+c_I$$
$$U_I'(k;o,o,o;k) = 0$$

Because of $U_j(s,l,a,p) = \int U_j'(s,l.a.p,t)dP_j(t|s,l,a,p)$ $(j=R,P,I)$, i.e. U_j is the subjective expected utility of the utility of the payoff:

$$U_j(k,o,o,o) = U_j(k,o,o,o,k) = 0 \quad (j = R,P,I) \quad ,$$

since the conditional probability $P_j(k|k,o,o,o)$ the transition from state k to state k occurs is one.

$$U_R(C,L,l,a,p) = \begin{cases} c_1 l + c_2 a + c_3 p, & \text{if } C+\beta a \le C_R \text{ or } C_R < C; \\ c_1 l + c_2 \frac{C_R - C}{\beta} + c_3 p + c_R, & \text{if } C \le C_R \; C+\beta a; \end{cases}$$

$$U_P(C,L,l,a,p) = c_4 a;$$

$$U_I(C,L,l,a,p) = \begin{cases} c_5 a + c_6 p, & \text{if } C+\beta a \le C_I \text{ or } C_I < C; \\ c_5 \frac{C_I - C}{\beta} + c_6 p + c_I, & \text{if } C \le C_I < C+\beta a \;. \end{cases}$$

The parameters are assumed to have the following signs $c_1 \ge o$, $c_2 > o$, $c_3 < o$, $c_4 > o$, $c_5 > o$, $c_6 > o$ whereas c_R, c_P, c_I are large negative payoffs. $c_1 \ge 0$ reflects the regulator's internal difficulties to set small standards, $c_2 > o$, $c_4 > o$, $c_5 > o$ the benefits of energy production, $c_3 < o$ the damage of pressure, and $c_6 < o$ the burden of organization. It turns out that these assumptions already determine the shape of the range of the payoffs.

A Multistage Model for Noise Problems

Since the opening of the fast railway line Shinkansen in 1964, complaints about noise and vibration have never ceased. Up to now the Japanese National Railways have been reluctant to take steps towards noise reduction such as building soundwalls, dislocation of neighbors, and slowing down trains. So far the impactee's mearsures have gone through all the possible stages: complaints, petition to the government, organization of citizens for negotiations with Japanese National Railways and the government, and legal proceedings. The regulator consists of various institutions (like the Environmental Agency, for example) with expert committees and subcommittees, local government, and national government. For a better understanding of the basic structure, the institutional aspects are neglected and the regulator is formalized as one player. The impactee is characterized by a response function.

The states of the game are a subset of

$$\{(L,i) \mid \underline{n} \le I \le \bar{n},\; i = 1,2,\ldots 7\} \quad ,$$

where L denotes an upper bound for an admitted noise level, $\bar{n}$ the maximum value of noise produced by the train operated only under economic considerations, and $\underline{n} > 0$ the minimum value of noise under which the train can be run under economic considerations. (L,1) is the first state after construction of the railway line.

Hence $(L,1) = (\bar{n},1)$. State $(L,2)$ indicates that a petition has been filed. $(L,3)$ states that the population affected by noise has organized itself to negotiate with the government for a low noise standard. If negotiations fail the impactee can start a lawsuit, which is indicated by $(L,4)$. $(L,4)$ can be followed by states of type $(L,5)$, $(L,6)$, or $(L,7)$. $(L,5)$ denotes that a permanent compromise has been achieved with upper bound L for noise: $(L,6)$ that the lawsuit was decided in a neutral or positive way for the Japanese National Railways and the government; and $(L,7)$ that the lawsuit was decided in favor of the impactee. $(L,5)$, $(L,6)$, and $(L,7)$ are final or absorbing states.

For each class of states the component game and the transition probability are given separately.

It is assumed that the costs and benefits of the train have aggregated such that the utility of the regulator is given as a function on the values of noise:

$$u_R : [\underline{n},\bar{n}] \to \mathbb{R} ,$$

as long as there is no action on part of the population. u_R is assumed to be unimodal, i.e. it is strictly increasing on $[\underline{n},L^+]$ and strictly decreasing on $[L^+,\bar{n}]$ where $L^+ \varepsilon [\underline{n},\bar{n}]$. u_R reflects a compromise among the economic importance of the train and the detrimental effect on the neighboring residents. As long as there is no regulation the (noise-) producer's utility is specified by the strictly increasing function

$$u_P : [\underline{n},\bar{n}] \to \mathbb{R} ,$$

based completely on economic considerations.

In the case of the first state $(L,1) = (\bar{n},1)$ the sets of choices are specified by

$$M_R(\bar{n},1) \quad = \{l|\underline{n} \le l \le \bar{n}\} ;$$

$$M_P(\bar{n},1,l) = \{n|\underline{n} \le n \le l\} ;$$

where l denotes the utmost level of noise allowed to the producer, and n the value of noise generated by railway operation. The impactee's choices are not specified since the impactee is formalized by a response function resulting in special transition probabilities.

Given state $(\bar{n},1)$ only states $(\bar{n},1)$ and $(\bar{n},2)$ can succeed. A critical noise level $n_I \varepsilon [\underline{n},\bar{n}]$ is assumed for the impactee such that noise is regarded as a substantial impact if and only if its value is greater than n_I. The subjective transition probabilities are specified by

$$P((\bar{n},2)\ \bar{n},1,l,n) = \begin{cases} 0 & \text{if } n \leq n_I ; \\ p_2 & \text{if } n > n_I ; \end{cases}$$

$$P((\bar{n},1)\ \bar{n},1,l,n) = \begin{cases} 1 & \text{if } n \leq n_I ; \\ 1-p_2 & \text{if } n > n_I . \end{cases}$$

Indices j = R,P for the subjective probabilities are omitted since this model assumes that regulator and producer consult the same experts. $p_2 > 0$ represents the experts subjective probability that the impactee will prefer a petition. The utilities are given by

$$\begin{aligned} U_R(\bar{n},1,l,n) &= u_R(n) ; \\ U_P(\bar{n},1,l,n) &= u_P(n) . \end{aligned} \qquad (\underline{n} \leq n \leq \bar{n})$$

Given state $(\bar{n},2)$ the set of measures are

$$M_R(\bar{n},2) = \{l \mid \underline{n} \leq l \leq \bar{n}\} ;$$

$$M_P(\bar{n},2,l) = \{n \mid \underline{n} \leq n \leq l\} .$$

Then $(\bar{n},2)$ can only be replaced by $(\bar{n},3)$ denoting the formation of an organization. We assume the following subjective transition probabilities:

$$P((\bar{n},3) \mid \bar{n},2,l,n) = \begin{cases} 0 & \text{if } n \leq n_I ; \\ p_3 & \text{if } n > n_I ; \end{cases}$$

$$P((\bar{n},2) \mid \bar{n},2,l,n) = \begin{cases} 1 & \text{if } n \leq n_I ; \\ 1-p_3 & \text{if } n > n_I ; \end{cases}$$

where $p_3 > 0$. The idea is that $n \leq n_I$ is conceived as giving in by either the regulator by $l \leq n_I$ or by the producer in the case of $n \leq n_I < l$. The payoffs are specified by

$$\begin{aligned} U_R(\bar{n},2;l,n) &= u_R(n) ; \\ U_P(\bar{n},2;l,n) &= u_P(n) . \end{aligned} \qquad (\underline{n} \leq n \leq l)$$

If an organization is formed (which is denoted by $\bar{n},3$) it is the impactee's objective to have the regulator give in.

Let

$$M_R(\bar{n},3) \quad := \{l \mid \underline{n} \le l \le \bar{n}\} \quad ;$$

$$M_P(\bar{n},3,l) := \{n \mid \underline{n} \le n \le l\} \quad ;$$

then

$$P((\bar{n},4)\mid\bar{n},3,l,n) = \begin{cases} 0 & \text{if } l \le n_I \quad ; \\ p_4 & \text{if } l > n_I \quad ; \end{cases}$$

$$P((\bar{n},3)\mid\bar{n},3,l,n) = \begin{cases} 1 & \text{if } l \le n_I \quad ; \\ 1-p_4 & \text{if } l > n_I \quad ; \end{cases}$$

where $p_4 > 0$ and $(\bar{n},4)$ denotes the start of a lawsuit. Let

$$U_j(\bar{n},3,l,n) := u_j(n) \qquad (j = R,P; \quad \underline{n} \le n \le \bar{n})$$

Three outcomes of a lawsuit are considered. There is a compromise (L,5) suspending the lawsuit, or a sentence in favor of regulator and producer (L,6), or a sentence in favor of the impactee (L,7). Let

$$M_R(\bar{n},4) \quad = \{l \mid \underline{n} \le l \le \bar{n}\} \cup \{(l,\Lambda) \mid \underline{n} \le l \le \Lambda \le \bar{n}\} \quad ;$$

$$M_P(\bar{n},4,l) = M_P(\bar{n},4,l,\Lambda) := \{n \mid \underline{n} \le n \le l\} \cup \{(n,N) \mid \underline{n} \le n \; \varepsilon \; N < \bar{n}, \; n \le l\} \; .$$

Let

$$M_C := \{(l,\Lambda;m_P) \mid (l,\Lambda) \; \varepsilon \; M_R(\bar{n},4), \; \Lambda \le n_I, \; m_P \; \varepsilon \; M_P)\bar{n},4,l)\}$$
$$\cup \; \{(m_R;n,N) \mid m_R \; \varepsilon \; M_R(\bar{n},4), \; (n,N) \; \varepsilon \; M_P(\bar{n},4,m_R), \; N \le n_I\}$$

be called the set compromise pairs of choices. Then we assume

$$P((L,5)\mid\bar{n},4,m_R,m_P,) = \begin{cases} 1 & \text{if } (m_R,m_P) \; \varepsilon \; M_C \text{ and } L = \min(\Lambda,N), \\ & \text{where } \Lambda := +\infty \text{ or } N := +\infty \text{ unless} \\ & \text{defined previously;} \\ 0 & \text{else;} \end{cases}$$

$$P((L,6)\mid\bar{n},4,m_R,m_P) = \begin{cases} p_6 & \text{if } L = n_R \text{ and } (m_R,m_P) \notin M_C; \\ 0 & \text{else;} \end{cases}$$

$$P((L,7)\mid\bar{n},4,m_R,m_P) = \begin{cases} p_7 & \text{if } L = n_I \text{ and } (m_R,m_P) \notin M_C; \\ 0 & \text{else;} \end{cases}$$

where $\underline{n} \leq n_I \leq n_R \leq \bar{n}$ for the maximal noise level n_R fixed by the court is in favor of the producer, $0 \leq p_6 + p_7 \leq 1$. Hence

$$P((\bar{n},4)|\bar{n},4,m_R,m_P) = \begin{cases} 0 & \text{if } (m_R,m_P) \in M_C \quad ; \\ 1-p_6-p_7 & \text{if } (m_R,m_P) \notin M_C \quad . \end{cases}$$

The payoffs are specified by

$$\begin{aligned} U_R(\bar{n},4,m_R,n) &= U_R(\bar{n},4,m_R,n,N) = u_R(n) \\ U_P(\bar{n},4,m_R,n) &= U_P(\bar{n},4,m_R,n,N) = u_P(n) \end{aligned} \qquad (\underline{n} \leq n \leq 1) \quad .$$

State (L,5) means that either the regulator has agreed to take $L \leq n_I$ as the maximal noise level or that the producer has bound himself to noise levels not higher than $L \leq n_I$.

Let

$$M_R(L,5) := \{1|\underline{n} \leq 1 \leq L\} \quad ,$$

$$M_P(L,5) := \{n|\underline{n} \leq n \leq 1\} \quad ,$$

then

$$P((L,5)|L,5,1,n) = 1.$$

The payoffs are specified by

$$U_j(1,5,1,n) = u_j(n) \qquad (j = R,P;\ \underline{n} \leq n \leq 1) \quad .$$

State $(n_R,6)$ indicates a sentence unfavorable to the impactee.

Let

$$M_R(n_R,6) = \{1|\underline{n} \leq 1 \leq n_R\} \quad ,$$

$$M_P(n_R,6,1) = \{n|\underline{n} \leq n \leq 1\} \quad ,$$

Then $P((n_R,6)|n_R,6,1,n) = 1$,

$$U_j(n_R,6,1,n) = u_j(n) \qquad (j = R,P) \quad .$$

State $(n_I,7)$ signifies a sentence unfavorable to regulator and producer. Let

$$M_R(n_I,7) = \{1|\underline{n} \leq 1 \leq n_I\},\ M_P(n_I,7,1) = \{n|\underline{n} \leq n \leq 1\} \quad .$$

Then

$$P((n_I,7)|n_I,7;1,n) = 1 \quad .$$

The payoffs are given by

$$U_j(n_I,7,1,n) = u_j(n)+c_j \qquad (j = R,P) \quad .$$

c_j 0 expresses the freedom of decisions lost for other industrial activities involving noise since the sentence must be taken into account for the designing of such activities. In the case of $L^+>n_I$, it is assumed that c_j is a negative multiple $-m_j$ of $u_j(L^+) - u_j(n_I)$, i.e.

$$c_j = -m_j(u_j(L^+) - u_j(n_I)) \quad .$$

Since $(L,5)$, $(n_R,6)$, and $(n_I,7)$ are permanent states the payoffs for plays will only exist for proper discount factors $\rho_R<1$ and $\rho_P<1$. ρ_R and ρ_P need not be equal. Sometimes $\rho_P<\rho_R$ seems to be an adequate assessment.

SOLUTION CONCEPTS

Given the strategy-sets Σ_j $(j = R,P,I)$ of the three players and the vector of utilities (V_R,V_P,V_I) defined on the cartesian product $\Sigma_R x \Sigma_P x \Sigma_I$ of the strategy sets, each player faces the problem of selecting a strategy in order to obtain a high utility. Features that have to be considered in the selection of appropriate strategies are precisely formulated as solution concepts. However, except for two-person zero-sum games, there is no unique solution concept for general n-person games (see R.D. Luce, H. Raiffa, 1957), (J.C. Harsanyi, 1977).

In the following we will introduce several familiar solution concepts and discuss their applicability to the problem of procedures for standard setting which depends on the specific structure of the conflict situation, and the purpose of our analysis.

Definition

A three-tuple $(\sigma_R^+,\sigma_P^+,\sigma_I^+) \in \Sigma_R x \Sigma_P x \Sigma_I$ of strategies is called a *weak equilibrium point* if

$$V_R(\sigma_R^+,\sigma_P^+,\sigma_I^+) \geq V_R(\sigma_R,\sigma_P^+,\sigma_I^+) \qquad (\sigma_R \in \Sigma_R) \quad ;$$

$$V_P(\sigma_R^+,\sigma_P^+,\sigma_I^+) \geq V_P(\sigma_R^+,\sigma_P,\sigma_I^+) \qquad (\sigma_P \in \Sigma_P) \quad ;$$

$$V_I(\sigma_R^+,\sigma_P^+,\sigma_I) \geq V_I(\sigma_R^+,\sigma_P^+,\sigma_I) \qquad (\sigma_I \in \Sigma_I) \quad .$$

The three-tuple $(\sigma_R^+, \sigma_P^+, \sigma_I^+)$ is called a *strong equilibrium* point if the left-hand term of an inequality is always larger than the right-hand term.

Discussion

Equilibrium points are points of stability inasmuch as no player can improve his payoff if all the players persist in their equilibrium strategy. There is no statement as to how to arrive at an equilibrium point. In R.D. Luca, H. Raiffa, (1957, p. 91), it is pointed out that it is advantageous in such a situation to disclose one's strategy first and to have a reputation for inflexibility. A further complication is that several equilibrium points can exist.

It can be proven that the j-th component of the equilibrium payoff vector $(V_R(\sigma_R^+, \sigma_P^+, \sigma_I^+), V_P(\sigma_R^+, \sigma_P^+, \sigma_I^+), V_P(\sigma_R^+, \sigma_P^+, \sigma_I^+))$ is at least as large as the corresponding maximum payoff which is defined as $\max_{\sigma_j} \inf_{\sigma_i (i \varepsilon \{R,P,I\} \setminus \{j\})} V_j(\sigma_R, \sigma_P, \sigma_I)$.

The following solution concept makes sense only if some collusion is possible.

Definition

Let $\mathfrak{U} \subseteq \mathbb{R}^3$ denote the range of the utility functions:

$$\mathfrak{U} = (x_1, x_2, x_3) \varepsilon \mathbb{R}^3 \quad x_j = V_j(\sigma_R, \sigma_P, \sigma_I)$$

for one $(\sigma_R, \sigma_P, \sigma_I) G \Sigma_R x \Sigma_P x \Sigma_I \}$. The payoff vector $(u_R, u_P, u_I) \varepsilon \mathfrak{U}$ is called *Pareto-optimal* if there is no $(v_R, v_P, v_I) \varepsilon \mathfrak{U}$ such that

$u_j \leq v_j$ $(j = R,P,I)$

and $u_j < v$ for one j at least.

Discussion

Pareto-optimal payoff vectors are the undominated payoff vectors. Usually they exist in abundance. They are important in the case of collusion because then one can expect the players to use strategies yielding Pareto-optimal payoffs.

So far no comparison of utilities has been necessary. This is different for the following concept.

Definition

Let (u_R^+, u_P^+, u_I^+) denote the point of maximal possible payoffs called *bliss point,* i.e.

$$u_j^+ = \max(u_j \mid (u_R, u_P, u_I) \in \mathfrak{U}) \quad (j = R,P,I) \quad .$$

The payoff vector (u_R, u_P, u_I) is called *bliss-optimal* if

$$\sum_{j=R,P,I} (u_j^+ - u_j)^2 = \min(\Sigma(u_j^+ - v_j)^2 \mid (v_R, v_P, v_I) \in \mathfrak{U}) \quad .$$

Discussion

The bliss-optimal point depends on the norm. Here we have chosen the euclidean norm, but it is quite obvious that an l^p-norm with $p \neq 2$ may give other results. Furthermore, if the utilities are changed by linear positive transformation, the new bliss-optimal point is only in special cases related to the former by the same utility transformations.

Although R.D. Luce and H. Raiffa (1957) point out that the following concept is independent of positive affine transformations, this is no longer true for more general transformations.

Definition

Let (d_R, d_P, d_I) be a triple of payoffs the players obtain if they cannot reach an unanimous agreement or the choice of a payoff vector $u \in \mathfrak{U}$. Then the *Nash solution* is the point (u_R^+, u_P^+, u_I^+) at which the term $(U_R - d_R)\ (U_P - d_P)\ (U_I - d_I)$ is maximized subject to the requirement $(u_R, u_P, u_I) \in \mathfrak{U},\ u_j \geq d_j\ (j = R,P,I)$.

Discussion

d_j are called conflict payoffs. It is obvious that a Nash solution is Pareto-optimal. By definition as a product, the term $\prod_{j=R,P,I} (u_m - d_j)$ gives the same weight to each utility, hence the Nash solution is symmetrically dependent on the utilities. Sometimes d_j is assumed to be the maximum payoff of player j.

So far concepts without special assumptions about the announcement of strategies have been discussed. The following deals with a leadership concept yielding a different solution concept. It is assumed that the regulator has to announce **his** strategy first and then the producer. Optimal responses on part of the impactee and the producer can be regarded as solutions.

Definition

A *hierarchic solution* is a three-tuple (τ_R, τ_P, τ_I) of a strategy $\tau_R \varepsilon \Sigma_R$, and two maps

$$\tau_P : \Sigma_R \to \Sigma_P \quad ,$$

$$\tau_I : \Sigma_R \times \Sigma_P \to \Sigma_I \quad ,$$

such that

$$U_I(\sigma_r, \sigma_p, \tau_i(\sigma_R, \sigma_p)) = \max_{\sigma_I \varepsilon \Sigma_I} V_I(\sigma_R, \sigma_P, \sigma_I) \quad ;$$

$$V_P(\sigma_R, \tau_P(\sigma_R), \tau_I(\sigma_R, \tau_P(\sigma_R)) = \max_{\sigma_P \varepsilon \Sigma_P} V_P(\sigma_R, \sigma_P, \tau_I(\sigma_R, \sigma_P)) \quad ;$$

$$V_R(\tau_R, \tau_P(\tau_R), \tau_I(\tau_R, \tau_P(\tau_R)) = \max_{\sigma_R \varepsilon \Sigma_R} V_R(\sigma_R, \tau_P(\sigma_R), \tau_I(\sigma_R, \tau_P(\sigma_R)) \quad .$$

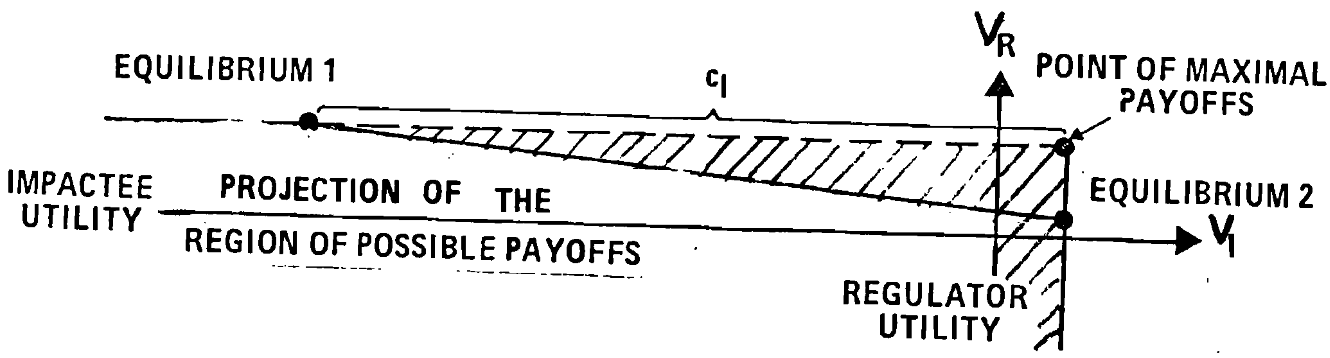
Discussion

The definition of hierarchic solution indicates that such a solution is the solution of a dynamic programming problem over function spaces. Hence, besides the rather restrictive requirements sufficient for the existence of a solution (K. Hinderer, 1970), the calculation of a solution can be carried out only for special models. However, the hierarchic solution is especially convincing if the corresponding payoffs are Pareto-optimal since then collusion cannot increase the payoff of all players. Furthermore, it is an equilibrium point, as can easily be seen. In the case of a one-stage game, the hierarchic solution coincides with the solution concept used in D.v. Winterfeldt, (1978, 1) under the conditions specified there.

CONCLUDING COMMENTS

Whether it is worth it or not to develop a game-theoretic framework in any sense (e.g. normative or descriptive) for the conflict situation among the interest groups involved in a pollution problem, can only be decided on the basis of case studies. Actually, there is only the study on carbon dioxide where all the solution concepts have been applied, and the noise study where due to a lack of time only the hierarchic solution was applied.

If in the case of carbon dioxide the impactee is more cautious than the regulator, a region of possible payoffs is that in the following Figure 2, assuming that the producer acts rationally.



Equilibrium 1: $\left(c_2 \frac{C_R - C}{\beta},\ c_5 \frac{C_R - C}{\beta} + c_I\right)$

Equilibrium 2: $\left(c_2 \frac{C_I - C}{\beta},\ c_5 \frac{C_I - C}{\beta}\right)$

Figure 2. Payoff Diagram for Regulator and Impactee ($C_R > C_I$)

The Pareto-optimal points Equilibrium 1 and Equilibrium 2 actually stem from equilibrium points. It is obvious that Equilibrium 2 is an approximation of the bliss-optimal point and the Nash solution. The hierarchic solution concept, however, yields Equilibrium 1 as payoff vector. From the formulas given below Figure 2, one can see the parameters that determine the solution. The analysis has yielded strategies of the impactee that can be taken as an assessment of a response function. This oversimplified model already confirms the dominating importance of the parameters C_R and C_I.

The noise study once more demonstrates that the framework is broad enough for a variety of cases. While in some cases extensions might be appropriate, it seems that there exist basic features of the pollution problem, the structuring of which would specialize the framework in greater detail, thus rendering it much more powerful. One such feature is the monitoring aspect or surveillance whether the producer operates within the standard. Since there is an analysis of this problem in D. v. Winterfeldt, 1978, 1, and since both authors have knowledge of the inspection problem (R. Avenhaus, 1977), (R. Avenhaus, E. Höpfinger, 1970), (E. Höpfinger, 1975), this problem has been postponed especially since the approach of M. Maschler, 1966, where the inspector announces his inspection strategy, can apparently be carried over without too many difficulties. One other aspect not fully treated is the way of modification of subjective probabilities if new data are available. For an introduction, we refer to M.H. DeGrout (1970), and T.S. Ferguson (1967).

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III.

APPLICATIONS OF DECISION AND GAME THEORETIC MODELS

MODELLING STANDARD SETTING DECISIONS: AN ILLUSTRATIVE APPLICATION TO CHRONIC OIL DISCHARGES*

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Abstract

This research memorandum presents an illustrative application of a three decision maker, one stage decision model to the problem of setting chronic oil discharge standards. It is the third paper in a series dealing with decision models and their applications to standard setting. The first paper (IIASA, RM-78-5) describes the problematique of oil discharge standard setting faced by the UK and Norwegian governments with the growing oil development in the North Sea. The second paper (IIASA, RM-78-7) provides the formal background of a decision theoretic model which was developed to aid regulatory agencies in standard setting tasks. The present paper is a first attempt to apply this model to a real world standard setting case.

The paper took the UK standard setting problem as it arose in 1975 as its starting point. It first describes the regulation problem in terms of sources, amounts and effects of chronic oil discharges from North Sea production platforms. Then it presents a three decision maker, one stage decision model for standard setting which encompasses the decision making of a regulator unit, a developer unit, and an impactee unit. The decision making of the developer unit is thought as being influenced by a standard through the possibility of detections of violations of the standard and subsequent sanctions. The decision making of the impactee unit is influenced by the operation and treatment decision of the developer unit. Each decision unit is modelled by a decision theoretic model including quantifications of uncertainties and values, determining optimal treatment responses to regulation alternatives, and utilities for all three units as a function of standards.

The application of the model is illustrative in character. Although some of the data used in the model were collected during two field studies in the UK and Norway, many quantifications are still hypothetical in the sense that they are no real assessments by the experts and decision makers, but rather reflect the author's perceptions of the decision maker's and experts' opinions and evaluations. Nevertheless the model results provide some basic insights in the standard setting problem for chronic oil discharges.

The model showed the strong influence of the uncertainty of equipment performance on the decision making of the developer as a response to a standard. Even risk neutral decision makers would make rather conservative treatment decisions. The model also shows the sensitivity of the precise definitions of sample

size, sampling period and exemptions on the developer's decision making. The model demonstrates clearly that a standard is not equal to a standard, that is, the numerical value of a standard tells only part of the story about its strictness in changing the developer's decision making.

Other model parameters, such as penalty functions, utility functions, etc., proved to be rather insensitive. Another model result is that there are many dominated standards, i.e., standards which are not better for any decision unit and worse for at least one as compared to other standards. This dominance effect is a result of the discreteness of the developer's (treatment) response to a continuous decision of the regulator.

The general conclusion of this model application is that such a formalization of a standard setting problem is feasible in principle, that the model results make sense, and that running the model gives interesting insights in the sensitivities of certain decision parameters. The model application also showed that especially in the modelling of the political evaluations of the regulator and the environmental modelling of the impactees much improvement is necessary to allow a trade-off analysis between the three decision making units.

Contents

Modelling Standard Setting Decisions: An Illustrative Application to Chronic Oil Discharges

THE PROBLEM OF SETTING CHRONIC OIL DISCHARGE STANDARDS

Offshore oil production brings not only economic benefits to the oil producing countries, but also increased risks and hazards of oil pollution into the seas. One major source of such oil pollution are production platforms with their risks of oil blowouts and their continual chronic or operational oily water discharges. Although accidental spills like the Santa Barbara blowout in the Pacific (see Straughan, 1971) or the Ekofisk blowout in the North Sea (see Fischer, 1978) are more dramatic and have more visible impacts on fish, birds, and beaches, chronic discharges may not be less dangerous. Such discharges continue over years and they involve the possibility of mortality or chronic toxicity of marine organism, long term effects on spawning behavior and changes in the ecological balance. Environmental agencies all over the world have therefore made attempts to limit these chronic oil discharges through appropriate regulations.

Oil emission standards which are set on the oil concentration in the discharged water are the most common regulatory tool to reduce oil pollution from production platforms in the seas. In a previous paper (Fischer and von Winterfeldt, 1978) the decision process was analyzed by which the UK and Norway set such chronic oil discharge standards for North Sea production platforms. The focus of that analysis was on the actors involved in the decision making, their conflicting goals and objectives, their decision alternatives and their information processing and evaluation strategies. During this largely descriptive and problem oriented analysis a decision theoretic model was developed with the aim to aid regulatory bodies in similar standard setting tasks. The present paper is an attempt to apply this model to the problem of chronic oil discharge standards.

The application presented in this paper is illustrative in character. Although some of the data used in the model were collected during two field studies in the UK and Norway on the problem of chronic oil discharge standards, many quantifications are still hypothetical in the sense that they are no real assessments by the decision makers and actors involved but rather reflect the author's perceptions of the actor's values and opinions. Using as a background the UK standard setting problematique as it arose around 1975, the model was developed and run to analyse chronic oil discharge standard setting in similar situations.

In the following section first the problem of chronic oil discharges, their amounts and their possible effects will be briefly reviewed following reports by the US Academy of Sciences (1975), the US Council of Environmental Quality (1974), and the Central Unit on Environmental Pollution of the UK (1976). Then the problem of regulating such discharges is discussed as it presented itself to the UK governmental agencies in 1975. In the next section the decision theoretic model will be presented following more detailed papers by von Winterfeldt (1978 a and b). The model application, sensitivity analyses, and a discussion of some major results will conclude the paper.

Sources, Amounts and Effects of Chronic Oil Discharges

Oily water discharges from production platforms have three main sources: production water, displacement water, and runoff water. Production water is pumped out of the oil field together with the crude oil and is discharged into the sea after a separation and possible further treatment process. Crude oil may contain as little as one percent production water in the early days of production, and as much as 30% in later stages if water injection systems are used to increase the reservoir pressure (CUEP, 1976). Before treatment production water may contain up to 3000 ppm (parts per million) of oil. In offshore oil storage tanks at production platform sites water displaces the oil and mixes to some degree with it through diffusion and wave action. This water can contain up to 300 ppm of oil. Runoff water from production decks mixes with oil residues and small oil spills and is collected at the production platform.

Water from these three sources is usually collected and treated through various oil-water separation devices ranging from simple gravity separation over filtering devices to chemical or biological treatment. Good treatment practice with present day technology can reduce the oil content in oily water down to 20-50 ppm. In spite of these relatively low concentration levels, chronic oil discharges from production platforms still pose a potential danger to the marine ecology. First of all, the separation processes are usually survived by the more toxic soluable hydrocarbon components. Secondly, the total amounts of discharged oil is by no means small. Fischer and von Winterfeldt (1978) estimated that approximately 2200 tons of oil per year will enter the North Sea at peak production periods through chronic oil discharges (as a comparison: accidental oil spills are expected to lie around 5000 tons/year).

The possible effects of these chronic oil discharges on the marine environment are mediated through the ambient distribution of oil concentrations which is created around the platform through the constant flow of oily water emissions. This distribution is a result of a complicated process of emission, diffusion and decay. A recent MIT study estimated that an area of 5.4 km^2 around a platform can be polluted at levels higher than .1 ppm (or 100 ppb - parts per billion) (Council of Environmental

Quality, 1974). At such concentration levels several environmental impacts may occur ranging from direct mortality of marine organisms which are exposed for a longer period to such concentration, over long range cumulative effects to slight behavioral changes in the marine ecology.

Marine biologists are not at all certain about the nature and extent of these impacts. Early reports (NATO Report, 1970) warned of the potential dangers of oil pollution including possibilities of bioaccumulation of carcinogen substances. Laboratory experiments showed that continuous exposure of fish and other marine organisms at levels in the low ppm range can lead to mortality and toxicity. Recent reports are more optimistic (US National Academy of Sciences, 1975; Mertens and Allred, 1977; Westaway, 1977; Morgan et al., 1974). These reports conclude that there is little evidence for bioaccumulation and that most field studies of chronically polluting platforms gave no evidence of fish kills, toxicity, or ecological changes except in highly sensitive estuarian areas with low currents and flows. Yet marine biologists warn of the more subtle long term and behavioral changes, and urge regulatory agencies to set strict standards for chronic oil discharges from production platforms.

The Regulation Problem

When oil development went underway in the North Sea the UK Prevention of Oil Pollution Act (1971) formed the legal basis to regulate offshore oil pollution. This law forbade any oily water discharges from offshore installations without exemption. Naturally the development of the oil fields posed a dilemma to the UK regulatory agencies, the UK Department of Environment and the Petroleum Engineering Division (PED) of the Department of Energy. The solution was initiated with the new Petroleum and Pipelines Act (1975) which allowed offshore oil operators to discharge oily water provided they used best practicable means for treatment. There were, however, no precise guidelines to determine what "best practicable means" meant, and under which precise conditions offshore operators should be exempted from the no discharge law.

At the same time other European countries began to push the UK to set uniform chronic oil discharge standards for North Sea production platforms. A study team in the Central Unit on Environmental Pollution was set up in the Department of Environment to analyze the problem of discharge standards. Based largely on costs and performance data this study concluded that:

> "Data suggest that an average oil concentration of 30-40 ppm is achievable with present technology. A maximum effluent oil concentration of 100 ppm should not be exceeded more than 2% of the time. New development of existing systems for use on platforms may be able to reduce the effluent oil

concentration to an average of 20 ppm, but these systems have not yet been fully evaluated." (CUEP, 1976, p.22).

Partly as a result of this study the agency responsible for offshore platform regulation, the PED of the Department of Energy, set a standard of an average oil concentration of 40 ppm for platforms with discharge volumes of more than 100,000 barrels a day (approximately 14,000 tons) of oily water, and a standard of 50 ppm for platforms with smaller discharge volumes. These standards were effectively applied since 1977.

The effects of such standards on the regulator himself, the offshore oil operator, and the impacted fishermen and ecologists will be analyzed in the following section by a decision theoretic model. The model was developed in order to examine the results of changing the standard, possible sanctions or the monitoring procedure. Since the offshore oil operators' response to such regulation is a crucial element in regulatory decision making, a decision making model for such responses was developed. The impactees' responses were not modelled and instead impactees were considered sufferers or beneficiaries of the regulator's and the operator's decisions.

The model was quantified by using data from the CUEP, equipment manufacturers, and the PED. Where sufficient data was lacking, best guesses were substituted and checked with experts. Value and utility functions were not assessed interactively with decision makers but they are based purely on the author's perceptions of the decision maker's values, making use of parametric analysis to accommodate a range of possible functional forms.

A DECISION THEORETIC MODEL FOR ENVIRONMENTAL STANDARD SETTING

To develop a decision theoretic model for solving a complex decision problem, three main steps have to be followed:

1. The problem has to be structured. Structuring includes a definition of alternatives and events which may influence consequences in the form of a decision tree or payoff matrix (see Raiffa, 1968), and an elaboration and operationalization of the decision making objectives by means of goal trees (see Keeney and Raiffa, 1976).

2. Uncertainties and value judgements have to be quantified. This quantification includes the construction of probability distributions over uncertain events (see Spetzler and v. Holstein, 1975), and the construction of utility or value functions over consequences (see Fishburn, 1965; Keeney and Raiffa, 1976).

3. Probabilities, values and utilities have to be aggregated mathematically to an overall evaluation of action alternatives. This includes the aggregation probabilities through probabilistic inference models (see Kelly

and Barclay, 1973), the aggregation of values and utilities through multiattribute utility models (see Keeney and Raiffa, 1976; von Winterfeldt and Fischer, 1975), and the aggregation of probabilities and utilities through expected utility models (see v. Neumann and Morgenstern, 1947; Savage, 1954; Fishburn, 1970).

The structuring of the problem provides the following formal elements of the decision problem:

1. A set of alternatives (courses of action) A with typical elements a, b ε A;
2. A set of possible events S with may influence the consequence of an action a, with typical elements s, tε S;
3. A set of operationalized objectives $C_1, C_2, \ldots, C_i, \ldots, C_n$ which will also be called attributes. Each C_i is assumed to be a subset of R^1. The product set $C = \underset{i=1}{\overset{n}{X}} C_i$ is called the consequence space. Elements $\underline{c}$, $\underline{d}$ are called consequences.

The quantification part of the analysis provides

1. A probability distribution p which assigns to each s ε S a probability $p(s|a)$; and a probability density function f which assigns to each element $\underline{c}$ in the consequence space a value $f(\underline{c}|a,s)$.
2. A value function v: $C \to R$ which expresses the decision maker's relative preferences among consequences.

As an aggregation model a modified expected utility model is assumed. An evaluation function over A is defined as:

$$U(a) = \int_C \left[\sum_{s\varepsilon S} p(s|a) \cdot f(\underline{c}|a,s) \right] \cdot h[v(c)] \, d\underline{c} \tag{1}$$

Where h is a risk transformation which expressed - loosely speaking - the decision maker's attitude towards taking or avoiding chances. In the present model application a special form of (1) will be used, in which f is degenerate, h is either linear or exponential, and v is additive. (1) can then be written as

$$U(a) = \sum_{s\varepsilon S} p(s|a) \cdot (\text{sign } \alpha) \exp\{\alpha \sum_{i=1}^{n} w_i v_i [c_i [a,s)]\} \tag{2}$$

for $\alpha \neq 0$ and

$$U(a) = \sum_{s\varepsilon S} p(s|a) \sum_{i=1}^{n} w_i v_i [c_i (a,s)] \tag{3}$$

for $\alpha = 0$.

Here $v_i[c_i(a,s)]$ is the single attribute value of attribute C_i which obtains if a is the act selected and s occurs. The w_i's are scaling parameters, and α is a risk parameter. The decision maker is risk averse of $\alpha<0$, risk neutral if $\alpha = 0$, and risk prone if $\alpha>0$. According to models (1) to (3) that alternative $a\varepsilon A$ should be selected which has the highest value of U. The assumption behind this model are spelled out in v. Winterfeldt (1978 b).

In environmental standard setting tasks the decision model becomes more complicated, since there is no individual decision maker, but rather several decision makers with often conflicting interests and opinions. In a previous study (Fischer and von Winterfeldt, 1978) three main decision units were identified which typically enter into standard setting decision making:

1. The regulator unit
2. The developer or producer unit
3. The impactee unit.

The regulator unit consists of people and institutions which have to set the environmental standard, monitor its compliance and sanction its violation. Such agencies are usually environmental agencies or international control agencies. The developer unit is defined as all those people and institutions whose decision making is restricted by the standards, typically industrial or development organizations. In the impactee unit all those people or groups are combined whose activities or perceptions are impacted upon by the development or industrial activities through pollution. The main idea of the standard setting model is to build model (1) for each of these three units in terms of their own actions, alternatives, relevant events, etc., and to link them through the logic of the standard setting problem. The resulting regulator-developer-impactee model has a structure which is schematically represented in Figure 1.

If the regulator decides on a particular standard r, the decision making of the developer will be influenced by the possibility of sanctions if his operations d do not meet the standard. That is, the standard generates together with possible monitoring and sanctions the danger of a detection of a violation with its associated costs. The model of the developer will then be able to determine an optimal response d(r) which may not be the response or the action he would have selected without r.

Through pollution and their adverse effects on health and well being the developer influences the decision making of the impactees. Impactees may decide to leave polluted areas, or to begin legal action against the developer or the regulator. The model of the impactees can determine the optimal impactee action a(d) as a response to the developer's action d.

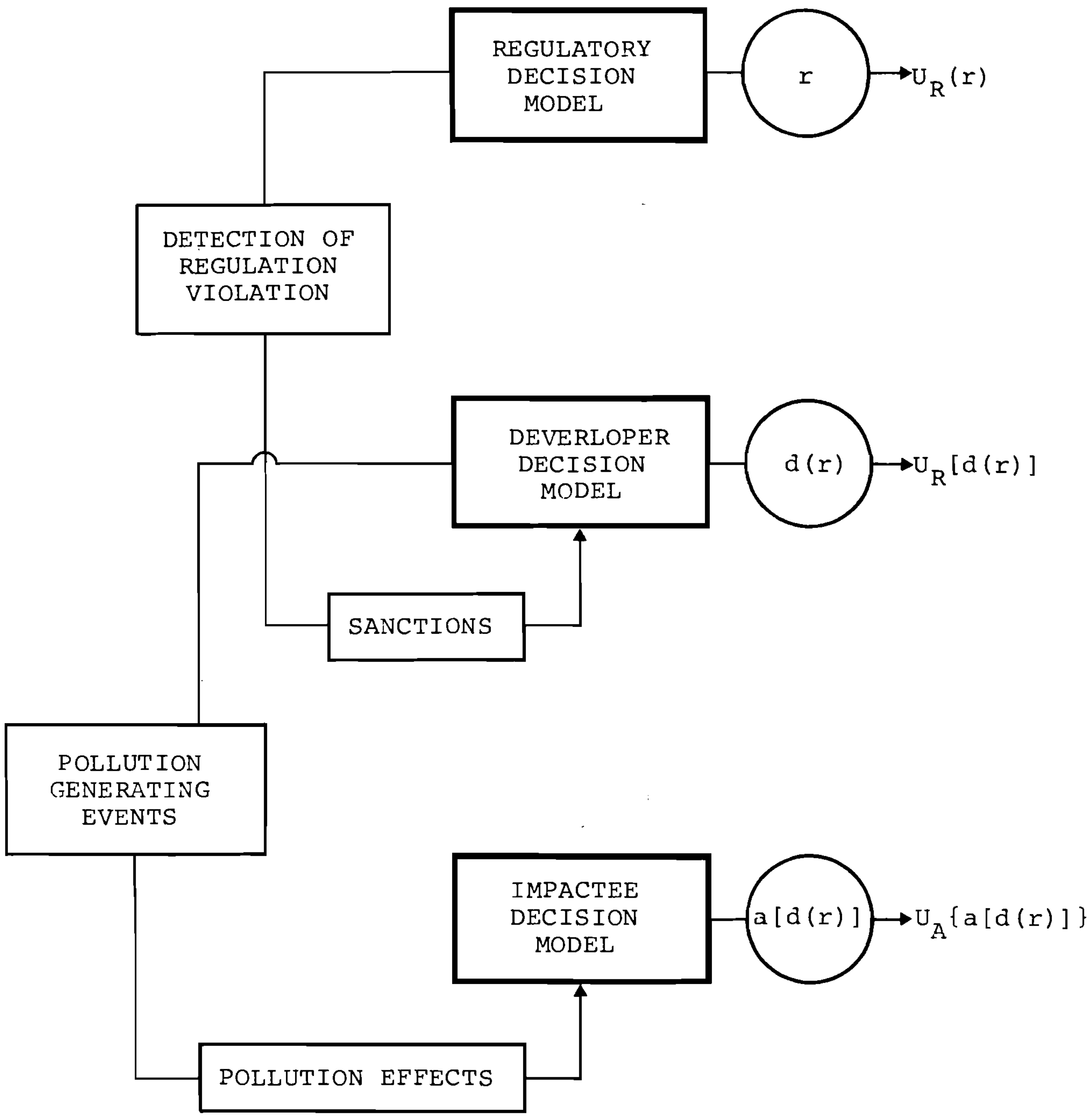

Figure 1 <u>SCHEMATIC REPRESENTATION OF THE REGULATOR-DEVELOPER-IMPACTEE MODEL</u>

The idea of the model is to determine optimal decisions d(r) and a[d(r)] for the developer and the impactee as a function of r together with the associated utilities U_R, U_D, and U_A. Further aggregation or a Pareto optimality analysis may then be used to focus in on a good value of r.

More formally, let Table 1 indicate the notation for the standard setting model.

Table 1 NOTATION FOR THE REGULATION MODEL

	SET	ELEMENT
Regulator's alternatives	R	r
Developer's alternatives	D	d
Impactee's alternatives	A	a
Regulator's consequences	C_R	$\underline{c}_R$
Developer's consequences	C_D	$\underline{c}_D$
Impactee's consequences	C_A	$\underline{c}_A$
Regulator's events	S_R	s_R
Developer's events	S_D	s_D
Impactee's events	S_A	s_A

The probability distributions and functions p_R, p_D, p_A, v_R, v_D, v_A, h_R, h_D, h_A are also assumed. Without specifying interlinkage models "detection of violations and sanctions" and "Pollution generation and effects" the evaluation of acts for each decision unit depends on the acts taken by other units, since $p_\bullet$ and $v_\bullet$ are dependent on all acts. To allow a separation, the following specifications and assumptions will be made:

1. $S_R = \emptyset$
2. $S_D = \{Q_0, Q_1\}$ where Q_0 stands for the event "non-detection of a violation" and Q_1 stands for "detection of a violation".
3. $S_A = \{l \mid 0 \leq l < \infty\}$ where l is the random emission level produced by the developer's activities.

Splitting R into three sets: the set of standard levels SL(sl$\in$SL), the set of monitoring and inspection devices SM(sm$\in$SM), and the set of sanctions SS(ss$\in$SS), the following utility functions U_R, U_D, and U_A are assessed:

$$U_R(r) = \sum_{i=1}^{n_R} w_{Ri} v_{Ri} [c_{Ri}(r)] \tag{4}$$

$$U_D(d,r) = \sum_{k=0}^{1} p_D(Q_k) \ (\text{sign } \alpha) \ \exp\{\alpha \sum_{i=1}^{n_D} w_{Di} v_{Di} [c_{Di}(d,ss,Q_k)]\} \tag{5}$$

for some $\alpha \neq 0$.

$$U_A(a,d) = \int_L f(l|d) \ (\text{sign } \beta) \ \exp\{\beta \sum_{i=1}^{n_A} w_{Ai} v_{Ai} [c_{Ai}(a,l)]\} \tag{6}$$

for some $\beta \neq 0$. For $\alpha, \beta = 0$, the linear form (3) applies.

The specifications and independence assumptions expressed in (3) to (5) can verbally be stated as follows:

(3) The regulator's consequences depend only on his own action;

(4) The developer's consequences depend only on his action, the possible sanctions, and the detection state.

(5) The impactee's consequences depend only on his actions and pollution levels.

This means, of course, that the U_R depends on r only, U_D on d and r only, and U_A on a and d only. Therefore d(r) and a(d) can be defined as follows:

$$d(r) : U_D(d(r),r) = \max_D U_D(d,r) , \tag{7}$$

$$a(d) : U_A(a(d),d) = \max_A U_A(a,d) . \tag{8}$$

(7) and (8) determine the optimal responses of the developer and impactee to a standard r and a development activity d respectively as well as the associated utilities as required. Therefore the model can determine the effects of r = (sl,sm,ss) on the three decision making units and by varying sl, sm and ss it can explore differential effects of monitoring procedures and sanctions. As a final analysis the model could be used to eliminate ordinally dominated standards and restrict further consideration to the Parato optimal set. The Parato optimal set can then be explored through weighting schemes of the form

$$\bar{U} = \lambda_R \cdot U_R + \lambda_D U_D + \lambda_A U_A . \tag{9}$$

THE REGULATOR MODEL

In this section the problem of setting chronic oil discharge standards as seen from the regulator's point of view will be structured, values will be quantified and some preliminary results concerning the regulator's utility function will be presented. In the sense of the standard setting model the regulatory agency in the UK which is concerned with setting chronic oil discharge standards is the Petroleum Engineering Division of the Department of Energy. The developer unit consists of the offshore oil operators, i.e., the large multinational oil companies and their contractors. Given an oil discharge standard, the offshore oil operators will have to adjust their production and treatment processes in order to meet that standard. The impactee unit is defined as the possible sufferers of chronic oil pollution, notably the fishery industry and possibly consumers of fish and other marine organisms. In addition, ecologists are considered impactees who are concerned about the more subtle behavioral and ecological changes in the marine environment.

The Structure of the Problem for the Regulator

Regulatory alternatives are the levels of oil emission standards, defined as the average amount of oil concentration in the effluent which is not to be exceeded. This set will be labelled SL (for standard level) with elements sl. SL has the following property:

$$SL = \{sl \mid 0 \leqslant sl < \infty\}$$

The procedure by which $\hat{l}$, the actual average oil concentration is determined is a matter of monitoring and inspection procedures SM. $\hat{l}$ is assumed to be computed as an average of n measurements taken over a fixed period of time t. Measurements are taken and analyzed by the operator himself, but periodical checks by inspectors are assumed to enforce "honest" reports.

As part of the monitoring and inspection procedure sm the regulator has to define what constitutes a detection of a regulation violation, i.e., when sanctions are applicable. For example, he may define such detection state as strictly as: "in continuous monitoring of the effluent the oil contration may <u>never</u> exceed the standard sl". As another example, he may consider a detection occurring only if the daily average of four samples exceeds sl more than twice in any given month during the lifetime of the plant. A distinction thus emerges between a single violation of the standard ($\hat{l}$>sl) and the detection state as defined here. In the definition of the monitoring and inspection procedure the regulator may decide to "overlook" a few cases of violations in a given time period, and only if the number of single violations becomes too large, a detection state would occur.

In general, detection states are defined as follows: the average $\hat{l}$ of n measurements taken during time period t should not exceed the standard sl more than m times during time period T(t<<T). For example, the 1978 US oil discharge standard (EPA, 1975) of 48 ppm is not to be exceeded by the average $\hat{l}$ of four samples (n = 4) taken daily (t = 1) more than twice (m = 2) in any one month (T = 30). SM thus is defined by {t,n,T,m}. The model will explore the following specific values, which roughly correspond to the US, UK, and Norwegian definitions of a maximum or an average standard*:

1. UK definition of a maximum standard (UK-MAX):

   Two samples are taken on every day. During any one month not more than two single samples (no averaging) may exceed the standard sl ($t = \frac{1}{2}$, n = 1, T = 30, m = 2)

2. EPA definition of an average standard (EPA-AV):

   The daily average of four samples may not exceed the standard sl more than twice during any one month (t = 1, n = 4, T = 30, m = 2)

3. Norwegian definition of an average standard (NWY-AV):

   The daily average of continuous sampling may not exceed the standard sl more than once during any one month (t = 1, $n \to \infty$, T = 30, m = 1)

4. UK definition of an average standard (UK-AV-0, UK-AV-1)

   The monthly average of two daily samples may not exceed the standard sl (more than once) during the lifetime of the plant (t = 30, n = 60, T = 5400, m = 0,1)

These definitions do not agree perfectly with the written definitions of the respective countries. For example, the EPA definition allows for 5% violations during any consecutive period of 30 days rather than two violations in one month and the UK definition of a maximum standard allows for 4% violations in any time period. Neither Norway nor the UK give firm values for the number of exemptions in their definition of an average standard. In the Norwegian case it seems clear that one exemption will be allowed, in the UK case both zero exemption and one exemption are analyzed.

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* In brackets: translation of regulation into the parameters of sm. A month is approximated by 30 days. The lifetime of the plant is assumed to be 180 30 day periods or approximately 15 years.

The regulator also has to determine sanctions, labelled SS. Sanctions are thought of as monetary penalty function, with the following form:

$$ss \equiv \begin{cases} 0 & \text{for } Q_0 \text{ (no detection)} \\ K_0 + K(d_j) & \text{for } Q_1 \text{ (detection)} \end{cases} \tag{10}$$

where K_0 is a fixed penalty and $K(d_j)$ is an additional penalty which depends on the treatment decision of the developer that led him to a violation of the standards. This additional penalty is interpreted as the cost necessary to improve the equipment and operations to meet the standard.

The regulatory objectives which the model will consider are all political in nature. According to the regulator the standard should:

(1) agree with standards of other nations

(2) satisfy international demands for a clean North Sea

(3) agree with national energy policy

(4) agree with national environmental policy.

Quantification for the Regulator Model

The degree to which the four regulatory objectives above are met will be expressed by functions on SL only. That means that C_{Ri} = SL and $c_{Ri}(r)$ = sl, i = 1,2,3,4. The first function v_{R1} is to express the degree to which sl agrees with standards of other nations. Four national standards were considered: Japan (10 ppm), France (20 ppm), Norway (approximately 40 ppm) and US (50 ppm). The value function is assumed to be bell shaped with a peak at the average of these four standards (30 ppm). Further restrictions assumed are:

$$v_{R1}(30) = 100 \tag{11}$$

$$v_{R1}(20) = v_{R1}(40) = 50 \tag{12}$$

$$v_{R1}(sl) \rightarrow 0 \quad sl \rightarrow \pm \infty . \tag{13}$$

The form selected was:

$$v_{R1}(sl) = 100 \cdot e^{-\frac{(sl-30)^2}{145}} . \tag{14}$$

The second value function v_{R2} is to express the degree to which sl meets international demands for a clean North Sea. An exponential function was selected with the following properties:

$$v_{R2}(0) = 100 \tag{15}$$

$$v_{R2}(50) = 50 \tag{16}$$

$$v_{R2}(sl) \rightarrow 0 \text{ for } sl \rightarrow \infty . \tag{17}$$

The final form selected was

$$v_{R2}(sl) = 100 \cdot e^{-.014 \cdot sl} . \tag{18}$$

The third value function should reflect the UK energy policy of rapid oil development in the North Sea. Clearly, the stricter the standard sl, the more likely it becomes that there will be a slow down of oil development. Therefore an exponential function was assumed with the following properties:

$$v_{R3}(0) = 0 \tag{19}$$

$$v_{R3}(50) = 50 \tag{20}$$

$$v_{R3}(sl) \rightarrow 100 \text{ for } sl \rightarrow \infty . \tag{21}$$

The form of that function is

$$v_{R3}(sl) = 100 - 100 \cdot e^{-.014 \cdot sl} , \tag{22}$$

The final value function should represent the UK environmental policy of using "best practicable means" for pollution abatement. This value function will build on a conclusion by the Paris Interim Commission on oil pollution that present day technology can achieve oil concentration reductions down to 40 ppm at reasonable cost. Since the UK agreed with this statement 40 ppm is considered achieveable with "best practicable means". The value function is assumed to be bell shaped around the value of 40 ppm with the following properties:

$$v_{R4}(40) = 100 \tag{23}$$

$$v_{R4}(20) = v_{R4}(60) = 50 \tag{24}$$

$$v_{R4}(sl) \to 0 \text{ for } sl \to \pm \infty . \tag{25}$$

The functional form is

$$v_{R4}(sl) = 100 \cdot e^{-\frac{(sl-40)^2}{577}} . \tag{26}$$

As can readily be seen, all v_{Ri}s are bounded between 0 and 100. To aggregate the v_i's weights w_i have to be assessed. These weights represent the importance which the regulator attaches to the four objectives. More precisely they are scaling factors which should reflect the relative degree of change in value by stepping from a standard with a value of 0 to one with a value of 100. Two weighting schemes will be explored in the model: a unit weighting scheme which attaches to all objectives the same weights, and a differential weighting scheme which seems to reflect more the actual importance attached to the objectives by the UK regulators:

Table 2 WEIGHTING SCHEMES FOR REGULATOR'S VALUE FUNCTION

	UNIT WEIGHTS	DIFFERENTIAL WEIGHTS
w_{R1}	.25	.10
w_{R2}	.25	.20
w_{R3}	.25	.40
w_{R4}	.25	.30

Some Preliminary Model Results for the Regulator

According to the model formulation of p. 9 the regulator's utility function U_R has the form

$$U_R(r) = \sum_{i=1}^{4} w_{Ri} v_{Ri}[c_{Ri}(r)] \tag{27}$$

which, in the above model quantification is defined as

$$U_R(sl) = \sum_{i=1}^{4} w_{Ri}v_{Ri}(sl) \tag{28}$$

with v_{Ri} and w_{Ri} specified as above.

$U_R(sl)$ is plotted in Figure 2 for the two weighting schemes. For the unit weighting scheme the best value of sl is 32 ppm, for the differential scheme 36 ppm. Thus both schemes do not differ much with respect to the optimal value of sl. In fact, the general shape of $U_R(sl)$ seems to be rather insensitive towards the weighting scheme as long as the weights are not too extreme. This is largely due to the fact that v_{R2} and v_{R3} "cancel" each other thus leaving most of the influence on U_R with the two bell shaped functions v_{R1} and v_{R4}. Therefore, for most weighting schemes "good" values of sl will lie in the range between 25 ppm and 45 ppm.

THE DEVELOPER MODEL

The Structure of the Problem for the Developer

The following treatment alternatives are possible responses of the developer to a standard level sl, a monitoring and inspection procedure sm and a sanction scheme ss:

d_1 : No treatment

d_2 : Simple gravity tank

d_3 : Corrugated plate interceptor (CPI)

d_4 : CPI and gas flotation (GF)

d_5 : CPI, GF, filtering (F)

d_6 : CPI, GF, F and biological treatment

d_7 : reinjection of oily water.

The details about these treatment alternatives are given in CUEP (1976). For the present purposes it is sufficient to say that d_1 to d_7 are ordered in increasing degrees of effectiveness in reducing oily water content in the effluent, and increasing costs. The model assumes that the developer has two conflicting objectives:

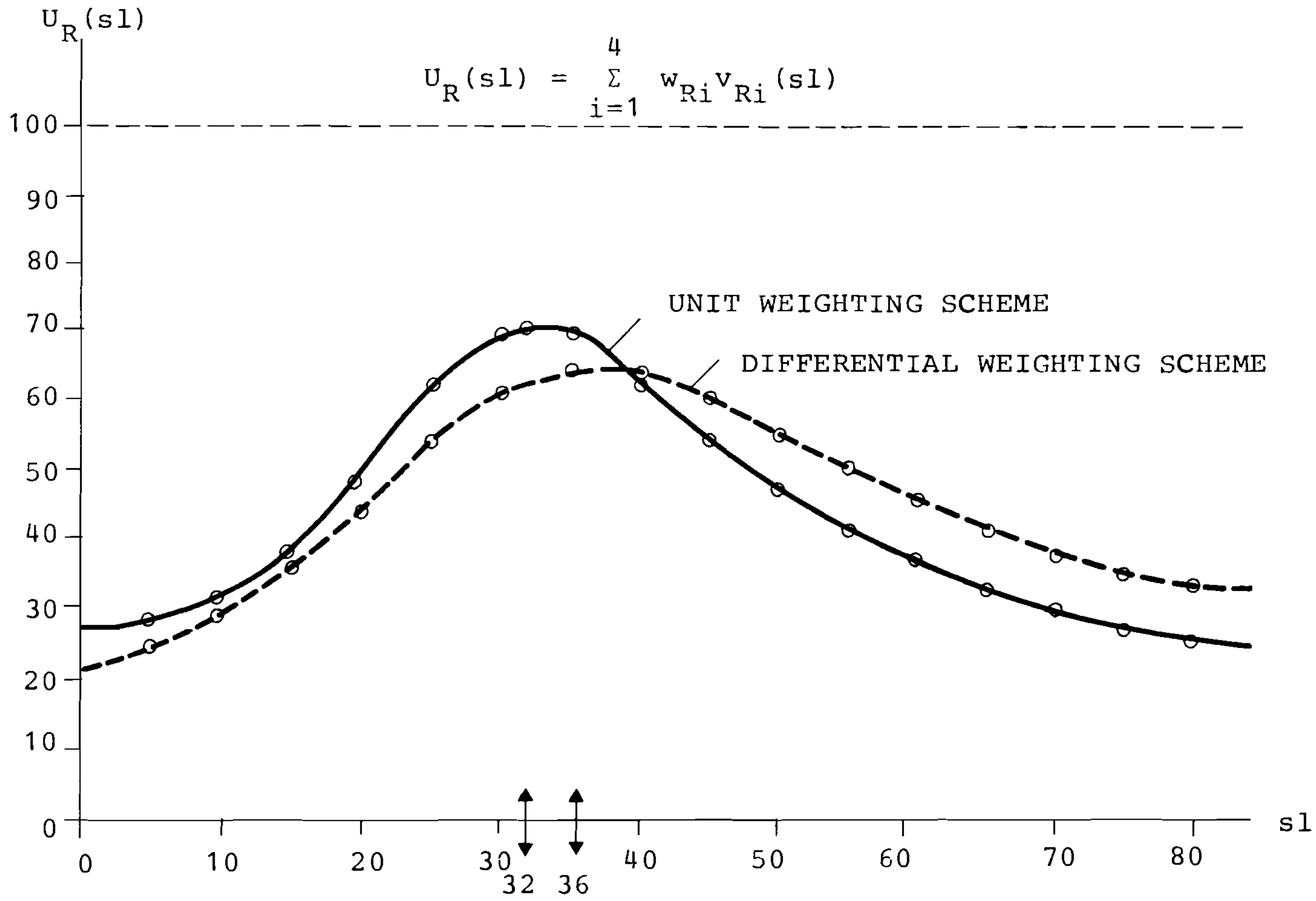

Figure 2 OVERALL VALUE FUNCTION OF THE REGULATOR FOR TWO WEIGHTING SCHEMES

(1) To minimize the cost of treatment, expressed in the attribute C_{D1} (in Pounds Sterling) with attribute values $c_{D1}(d_j,ss,Q_k)$ for $j = 1,\ldots,7$, $k = 0,1$.

(2) To minimize the penalties due to a violation of the standard, expressed in the attribute C_{D2} (in Pounds Sterling) with attribute values $c_{D2}(d_j,ss,Q_k)$, $j = 1,\ldots,7$, $k = 0,1$.

C_{D1} clearly depends on d_j only. Therefore

$$c_{D1}(d_j,ss,Q_k) = c_{D1}(d_j) \; . \tag{29}$$

However, violation costs are dependent on ss, Q_k and d_j. According to the penalty scheme (10)

$$c_{D2}(d_j,ss,Q_k) = \begin{cases} 0 & \text{if } k = 0 \text{ (no detection)} \\ K_0 + K(d_j) & \text{if } k = 1 \text{ (detection)} \end{cases} \tag{30}$$

$K(d_j)$ is interpreted as the cost of improving treatment on the request of the regulator. The model assumes that the regulator asks the developer to build in a treatment d_{j+1} if there is a detection of a violation with equipment d_j. $K(d_j)$ is defined as the additional cost

$$K(d_j) = c_{D1}(d_{j+1}) - c_{D1}(d_j), \; j = 1,\ldots,6 \; . \tag{31}$$

Therefore

$$c_{D2}(d_j,ss,Q_1) = K_0 + c_{D1}(d_{j+1}) - c_{D1}(d_j) \; . \tag{32}$$

(This assumption is realistic for d_1, d_3, d_4, and d_5, since the next best treatment just adds a unit to the already existing unit. For d_2 and d_6 the next best treatment is qualitatively different. Yet it is probably fair to assume that d_2 and d_6 would lead to savings when installing d_3 and d_7 after a violation, which corresponds to the outlays of d_2 and d_6).

Since both consequences c_{D1} and c_{D2} are expressed in monetary units, it is reasonable to assume that the value function V_D is additive and negative in cost:

$$-v_D(\underline{c}_D) = c_{D1} + c_{D2} \; . \tag{33}$$

Therefore

$$-v_D[\underline{c}_D(d_j, ss, Q_0)] = c_{D1}(d_j) + 0 \tag{34}$$

and

$$-v_D[\underline{c}_D(d_j, ss, Q_1)] = c_{D1}(d_j) + c_{D1}(d_{j+1}) - c_{D1}(d_j) + K_0 =$$

$$= c_{D1}(d_{j+1}) + K_0 \tag{35}$$

for $j = 1, \ldots, 6$.

The decision problem which the developer faces can therefore be expressed in a payoff matrix as in Table 3.

Table 3

PAYOFF MATRIX REPRESENTING THE DEVELOPER'S DECISION PROBLEM

		DETECTION STATES	
		NO DETECTION (Q_0)	DETECTION (Q_1)
ALTERNATIVES	d_1	$c_{D1}(d_1)$	$c_{D1}(d_2) + K_0$
	d_2	$c_{D1}(d_2)$	$c_{D1}(d_3) + K_0$
	d_3	$c_{D1}(d_3)$	$c_{D1}(d_4) + K_0$
	d_4	$c_{D1}(d_4)$	$c_{D1}(d_5) + K_0$
	d_5	$c_{D1}(d_5)$	$c_{D1}(d_6) + K_0$
	d_6	$c_{D1}(d_6)$	$c_{D1}(d_7) + K_0$
	d_7	$c_{D1}(d_7)$	-*

* No detection can occur for d_7.

To make the further analysis of the problem tractable, two important assumptions have to be introduced:

(1) It does not matter to the developer, when the detection occurs, that is a detection in the first year is not worse than a detection in the second, third, and so on.

(2) After detection, and after improvement of the equipment and payment of the penalty, no further detections are possible.

The first assumption is reasonably only if the developer does not discount future costs. The second assumption is reasonable if the next best treatment which the regulator asks the developer to build in after detection is so much better than the original treatment that further detection probabilities are essentially zero. In the sensitivity analysis of the developer model it will be shown that the second assumption is acceptable for the quantifications selected in the model.

With these assumptions cutoff probabilities of detection can be calculated at which the developer would switch from d_j to d_{j+1}. The developer's utility function was defined in (5) as

$$U_D(d,r) = \sum_{k=0}^{1} p_D(Q_k)(\text{sign } \alpha) \exp\{\alpha \sum_{i=1}^{n_D} w_{Di}v_{Di}[c_{Di}(d_j,ss,Q_k)]\} \tag{36}$$

which under the present specifications becomes

$$\begin{aligned} U_D(d_j,r) = {} & p_D(Q_0|d_j,sl,sm)(\text{sign } \alpha) \exp\{-\alpha c_{D1}(d_j)\} + \\ & + p_D(Q_1|d_j,sl,sm)(\text{sign } \alpha) \exp\{-\alpha[c_{D1}(d_{j+1}) + K_0]\} \end{aligned} \tag{37}$$

or in linear form

$$\begin{aligned} U_D(d_j,r) = {} & p_D(Q_0|d_j,sl,sm)(-1)c_{D1}(d_j) + \\ & + p_D(Q_1|d_j,sl,sm)(-1)[c_{D1}(d_{j+1}) + K_0]. \end{aligned} \tag{38}$$

The cutoff probability of detection at which the developer would switch from d_j to d_{j+1} is fully determined by the equation

$$U_D(d_j,r) = U_D(d_{j+1},r) \tag{39}$$

where, according to the assumptions above near the cutoff point

$$p_D(Q_0|d_{j+1},sl,sm) \cong 1. \tag{40}$$

Let this cutoff probability be labelled $p_D^*(d_j)$. Solving (39) for p_D^* gives

$$p_D^*(d_j) = 1 - \frac{K_0}{K_0 + c_{D1}(d_{j+1}) - c_{D1}(d_j)} \tag{41}$$

for the linear utility function, and

$$p_D^*(d_j) = 1 - \frac{\exp\{-\alpha[c_{D1}(d_{j+1}) + K_0]\} - \exp\{-\alpha\ c_{D1}(d_{j+1})\}}{\exp\{-\alpha[c_{D1}(d_{j+1}) + K_0]\} - \exp\{-\alpha\ c_{D1}(d_j)\}} \tag{42}$$

for the exponential form of the utility function.

The next step is to structure the probabilistic relationships between random emissions l, averages $\hat{l}$ and detection states Q_0 and Q_1. Following the model developed in v. Winterfeldt (1978) the assumption is made that each treatment d_j has performance characteristics in reducing the oil concentration in oily water which can be described by a normal distribution over levels l with mean l_j and standard deviation s_j. The averages $\hat{l}$ will therefore also have a normal sampling distribution with mean l_j and standard deviation $s_j/\sqrt{n}$. Let $p_o(d_j,sl,n)$ denote the probability that the average of n samples taken from the effluent treated with equipment d_j will not exceed the standard sl. Let $N(l_j,s_j)$ denote the normal probability distribution which characterizes the performance of equipment d_j. $N(l_j,s_j/\sqrt{n})$ therefore characterizes the sampling distribution of $\hat{l}$. p_o can be determined by standard methods through

$$p_o(d_j,sl,n) = \int_{-\infty}^{sl} N(l_j,s_j/\sqrt{n})\ d\hat{l}\ . \tag{43}$$

The calculation of detection probabilities $p_D(Q_k|d_j,sl,sm)$ will depend on the values of t (sampling period), T (inspection interval on which detection is defined), and m (number of exemptions from detection state), all of which are part of SM. Let M = T/t characterize the number of days in which violations ($\hat{l}$>sl) can occur: e.g., if T = 30 and t = 1 (in the EPA-AV scheme of page 11) there are 30 opportunities to establish $\hat{l}$>sl. The lifetime of the plant is assumed to be 180 30 day periods. Accordingly there will be N = 180 inspection intervals, if T = 30 and N = 1 inspection interval if T = 5400.

A detection is defined as m + 1 or more violations ($\hat{l}$>sl) during any one inspection interval T. The probabilities of i = 0,1,...,M violations during T follow a binomial distribution:

$$\text{Prob[i violations} \mid p_o, M] = \binom{M}{i} (1-p_o)^i \; p_o^{M-i}. \tag{44}$$

Therefore the probability p_m of no detection during any one inspection interval T is

$$p_m(d_j, sl, n) = \sum_{i=0}^{m} \binom{M}{i} (1-p_o) i \; p_o^{M-i}. \tag{45}$$

For example, in the EPA-AV scheme of page 11, where t = 1, T = 30, and m = 2, p_m is the probability of none, one or two violations during a 30 day period:

$$p_2(d_j, sl, sm) = p_o^{30} + 30p_o^{29}(1-p_o) + \frac{30 \cdot 29}{2} p_o^{28}(1-p_o)^2. \tag{46}$$

Given p_m, the probability of no detection over the lifetime of the plant can be determined as

$$p_D(Q_0 \mid d_j, sl, sm) = p_m^N. \tag{47}$$

In EPA's monitoring and inspection example N would be 180.

To give an example for a specific standard consider equipment d_j with l_j = 100 ppm and s_j = 8 ppm. Let sl = 110 ppm, and sm be the EPA-AV scheme. For this scheme the following results obtain:

p_o = .9938

p_2 = .9992

p_D = .8658

Often the situation is complicated, however, since the developer does not know the true average performance l_j, although he may know s_j for a given l_j. In fact, in the study by Fischer and v. Winterfeldt (1978) the uncertainty about treatment performance was considered by the oil industry a major factor influencing their decision making. To express this uncertainty about l_j a judgmental probability distribution is assumed over

l_j which is also normal with mean $\bar{l}_j$ and standard deviation $\bar{s}_j$. s_j is assumed to be known. Since l_j is distributed normally, and since the distribution of $\hat{l}$ can be interpreted as a conditional distribution given l_j, the marginal distribution of $\hat{l}$ is also normal with mean $\bar{l}_j$ and standard deviation

$$\hat{s}_j = \sqrt{\bar{s}_j^{\,2} + s_j^{\,2}/n} \; . \tag{48}$$

That means that the marginal distribution of $\hat{l}$ is equal to the sample distribution if there is no uncertainty about l_j. It is equal to the marginal distribution of l_j if there is uncertainty about l_j, but n is very large. In all cases in between it is a "dilluted" distribution in which uncertainty about the true mean performance l_j and the uncertainty about sampling are combined.

The values of $p_D(Q_k|d_j,sl,sm)$ can now be calculated from $\bar{l}_j$, $\bar{s}_j$, and s_j as follows:

(1) Given n, determine the marginal distribution of $\hat{l}$ which is $N(\bar{l}_j, \hat{s}_j)$.

(2) From $N(\bar{l}_j,\hat{s}_j)$ and sl determine p_o, the probability of no violation ($\hat{l} \leqslant sl$) during t.

(3) From p_o, m and T, determine p_m, the probability of no detection (less than m + 1 violation) during T.

(4) From p_m and N calculate $p_D(Q_0)$, the probability of no detection during the lifetime of the plant.

Another important analysis of the problem reverses these steps. Given a desirable low detection probability $p_D^*(d_j)$, one can approximate $p_o^*(d_j)$ the corresponding probability of no violation during any period t. From $p_o^*(d_j)$ and $N(\bar{l}_j,\hat{s}_j)$ a value $l_o^*(d_j)$ can be set for which this desirable low detection probability p_D would be achieved with d_j if $sl = l_0^*(d_j)$. Through these calculations cutoff levels $l_0^*(d_j)$ can be determined which correspond to the cutoff probabilities $p_D^*(d_j)$ (41, 42).

Quantification for the Developer Model

To quantify the costs for treatment and regulation violation which the developer faces, a reference production platform approach is selected. The assumption is made that the standard is to be set for a typical North Sea production platform with a daily oily water effluent of 10,000 tons. Also it is assumed that the platform is of a concrete type so that gravity separation and biological treatment are both technically feasible.

Since both attributes C_{D1} (cost of treatment) and C_{D2} (cost for violating the standard) are expressed in monetary units, the quantification for the developer is relatively easy. Table 4 lists some rough estimates of the possible costs for the seven options of the developer. These costs are based on some data by the Central Unit on Environmental Pollution (CUEP, 1976), on communication with manufacturers, and on a report of the National Academy of Sciences to the EPA (NAS, 1977). Yet, these cost estimates should be considered as merely illustrative of the model application. Total costs were calculated as the sum of installation cost and (undiscounted) operation costs for a fifteen year lifetime of the production platform.

Table 4

ILLUSTRATIVE COST ESTIMATES FOR DEVELOPER'S TREATMENT OPTIONS

(10^3 Pound Sterling); 1978

	Installation	Operation/yr.	C_{D1} Total (no disc.)
d_1 (no treatment)	0	0	0
d_2 (gravity tank)	0	3	45
d_3 (corr. plate int.)	70*	5	145
d_4 (CPI + GF)	140**	10**	290
d_5 (CPI + GF + F)	200	15	425
d_6 (CPI + GF + F + bio)	500	50	1250
d_7 (reinjection)	2100***	115***	3825

* Source: CUEP, 1976
** Source: Manufacturer's data
***Source: NAS, 1977 (for new platforms, far offshore)

All other data are rough estimates.

The sanction value K_0 is considered a parameter in the model. The following specific values will be analyzed: 10 000, 100 000, and 1 Mio. Pound Sterling. With this information the payoff matrix of page 18 can be filled for the sanction scheme:

Table 5

PAYOFF MATRIX FOR THE DEVELOPER

(entries in 10^3 Pounds, K_0 = 100,000 Pounds)

	DETECTION STATES	
	NO DETECTION (Q_0)	DETECTION (Q_1)
d_1	0	45 + 100
d_2	45	145 + 100
d_3	145	290 + 100
d_4	290	425 + 100
d_5	425	1250 + 100
d_6	1250	3825 + 100
d_7	3825	-*

* No detection is possible for d_7.

As inputs for the probability analysis estimates for the normal distribution characterizing the developer's uncertainty about the average equipment performance of d_j have to be made. These distributions are fully characterized by the mean $\bar{l}_j$ and standard deviation $\bar{s}_j$. In addition the standard deviation s_j of equipment performance for a given mean l_j has to be estimated.

Table 6 lists the estimates for $\bar{l}_j$, $\bar{s}_j$, and s_j which were made for the purpose of this model. The first column shows, in addition, some ranges of literature estimates of average equipment performance. Table 7 lists the mean and standard deviations of the marginal of $\hat{l}$ for n = 1,4,60, and ∞. These means and standard deviations characterize then the uncertainty about daily readings l for the different treatment options. From these distributions p_o, p_m, and p_D can be calculated as on p. 21 simply by substituting the "diluted" distribution of $\hat{l}$ for the precise sampling distribution which can be used if l_j and s_j are known. Figure 3 shows an illustration of the distributions in Table 7.

Table 6

LITERATURE RANGES OF AVERAGE TREATMENT PERFORMANCE AND PARAMETERS OF DISTRIBUTIONS CHARACTERIZING THE DEVELOPER'S UNCERTAINTY

TREATMENT	RANGE OF AVERAGE PERFORMANCE (ppm)* $l_{j,min} - l_{j,max}$	NORMAL DIST'N OVER l_j $\bar{l}_j$	$\bar{s}_j$	PERFORMANCE STANDARD DEV. s_j
d_1 (none)	300 - 3000	500	100	100
d_2 (Gravity)	50 - 150	100	20	20
d_3 (CPI)	∿50	50	5	10
d_4 (CPI + GF)	15 - 35	20	5	5
d_5 (CPI,GF,F)	3 - 10	5	2	3
d_6 (CPI,GF,F,B)	-	1	.2	.4
d_7 (reinject)	-	0	0	0

* Source: CUEP (1976) and equipment manufacturers.

Table 7

CALCULATED MEANS AND STANDARD DEVIATIONS OF AVERAGES $\hat{l}$ REFLECTING THE DEVELOPER'S UNCERTAINTY ABOUT THE TRUE PERFORMANCE OF THE TREATMENT EQUIPMENT

(Calculations based on Table 6)

TREATMENT d_j	MEANS $\bar{l}_j$	STANDARD DEVIATIONS $\hat{s}_j$			
		n = 1	n = 4	n = 60	n → ∞
d_1	500	141	112	101	100
d_2	100	28	22	20	20
d_3	50	11	7.1	5.2	5.0
d_4	20	7.1	5.6	5.1	5.0
d_5	5	3.6	2.5	2.0	2.0
d_6	1	.45	.28	.21	.20
d_7	0	0	0	0	0

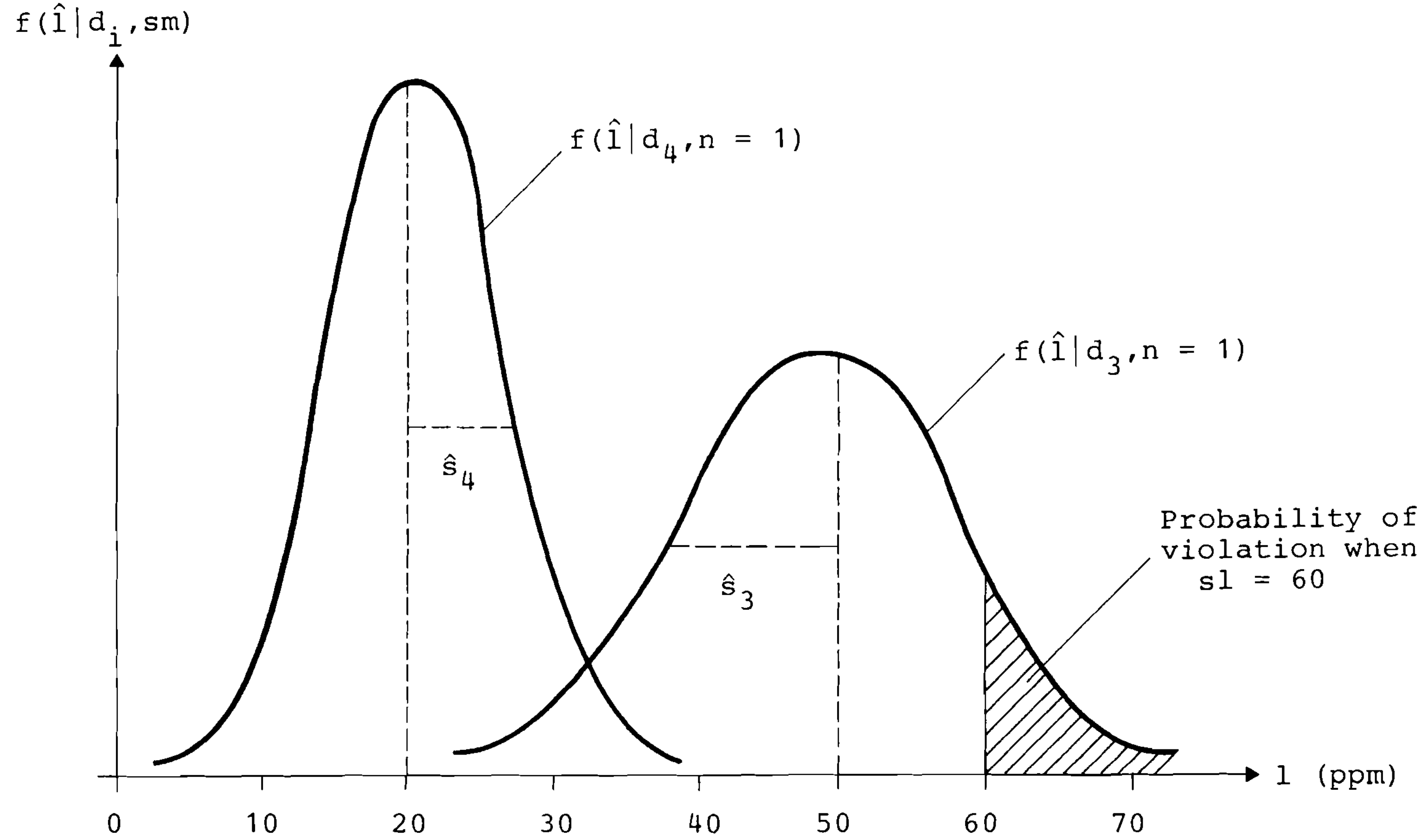

Figure 3 <u>ILLUSTRATION OF TWO NORMAL DISTRIBUTIONS CHARACTERIZING TREATMENT PERFORMANCE UNCERTAINTY</u>

Some Results of the Developer Model

The first results are the cutoff probabilities based on the payoff matrix in Table 5 and the formulas for cutoff probabilities p_D^* (41,42). Table 8 lists the cutoff probabilities for K_0 = 10 000, 100 000 and 1 Mio. Pound Sterling and for $\alpha = 0$, $2\cdot 10^{-6}$, $-2\cdot 10^{-6}$, i.e., for a risk neutral a risk prone, and a risk averse utility function.*

From the cutoff probability equations (41) and (42) it is already clear the p_D^* would be larger for small values of K_0 and smaller for large values. This trend is clearly represented in Table 8. Furthermore, for fixed K_0 the cutoff probabilities become larger the larger the cost differential between d_{j+1} and d_j. For a small penalty of K_0 = 10 000 Pounds the probability of detection must be in the .80 to .90s in order to force the developer model to switch to better treatment. But for the high penalty of 1 Mio. Pounds detection probabilities may be as low as .05 to have the model change from d_j to d_{j+1}.

The effect of the utility function is not very large for the two lower penalties. But, as one might expect, if potential losses of 1 Mio pounds are involved the shape of the utility function has a stronger influence on the cutoff probabilities. For K_0 = 1 000 000 it clearly shows that the model with the risk prone utility function would accept relatively high probabilities of detection before switching (for the switch from d_6 to d_7 even .995) while the risk averse utility function forces the model to switch at much lower probabilities (for the switch from d_6 to d_7 .135). In general, of course, all corresponding probabilities decrease from the risk prone over the risk neutral to the risk averse utility function.

From the cutoff probabilities of detection $p_D^*(d_j)$ in Table 8 the corresponding cutoff values $l_0^*(d_j)$ can be determined at which the developer would switch from treatment d_j to d_{j+1}. In terms of the standard sl these cutoff values divide the possible levels of sl into segments where each segment corresponds to a particular (optimal) choice of d_j; i.e., if the standard is just

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* To interpret the values of α, consider the gamble in which the decision maker would lose 1 Mio Pounds with a probability of .5 and nothing with probability .5. The risk neutral decision maker ($\alpha = 0$) would be willing to pay .5 Mio. Pounds to insure himself against the risk of the gamble, the risk prone decision maker ($\alpha = 2\cdot 10^{-6}$) would pay only .24 Mio Pounds, the risk averse decision maker ($\alpha = -2\cdot 10^{-6}$) would, on the other hand, pay .72 Mio Pounds.

Table 8

CUTOFF PROBABILITIES OF DETECTION $p_D^*(d_j)$ AT WHICH THE DEVELOPER WOULD SWITCH FROM d_j TO d_{j+1}

	RISK PRONE UTILITY FUNCTION ($\alpha = 2 \cdot 10^{-6}$)			RISK NEUTRAL UTILITY FUNCTION ($\alpha = 0$)			RISK AVERSE UTILITY FUNCTION ($\alpha = -2 \cdot 10^{-6}$)		
$K_0 (10^3)$	10	100	1000	10	100	1000	10	100	1000
d_1 (no)	.826	.342	.098	.818	.310	.043	.776	.268	.013
d_2 (Gravity)	.918	.550	.204	.909	.500	.091	.905	.456	.028
d_3 (CPI)	.944	.650	.280	.935	.591	.127	.926	.532	.038
d_4 (CPI,GF)	.937	.631	.264	.931	.574	.119	.921	.515	.036
d_5 (CPI,GF.F)	.995	.959	.829	.988	.892	.452	.976	.785	.114
d_6 (CPI,GF,F,B)	.999	.999	.995	.996	.962	.720	.980	.818	.135
d_7 (Reinject)	-	-	-	-	-	-	-	-	-*

* No detection is possible for d_7.

equal to $l_0^*(d_j)$, then the developer would be indifferent between choosing d_j or d_{j+1}. If the standard sl is tightened up ($sl<l_0^*(d_j)$), then the developer would select d_{j+1}; if sl is relaxed, he would prefer d_j.

Table 9 lists the values of l_0^* for the UK definition of an average standard (m = 1) and for different values of K_0 and α. This table reflects the picture of the detection probabilities in Table 8. Within each utility function block the cutoff levels become larger as the penalty K_0 increases. This effect is intuitively understandable and one would expect the model to behave in this way. The larger the possible penalty, the more cautious the developer would be, thus pushing up his cutoff values at which he would switch from inferior to improved treatment. This effect is clearest for the risk averse utility function and for the treatment options with a high uncertainty (i.e., none or gravity separation). Here the cutoff level shifts as much as 100 ppm (from 717 to 812) when the penalty is increased from 10 000 to 1 Mio Pounds. For the other treatment options the effect is much less dramatic.

A similar, although smaller effect of risk aversion vs. risk proneness can be seen. The more risk averse utility function leads to larger cutoff values. Again this is an affect which one would expect from the decision model: the more risk averse, the more cautious the cutoff levels should be set. This effect is only slight, although the choices of utility functions cover quite a substantial range in terms of the behavioral consequences.

Table 10 lists the cutoff values for the risk neutral utility function and K_0 = 100 000 for all five monitoring and inspection procedures. There are several noteworthy characteristics of the cutoff levels in Table 10. First, it shows that the definition of the monitoring and inspection procedure as part of the regulation has an effect on the decision making of the developer which is similar if not stronger than the effect of changing penalties or utility functions. For example $l_0^*(d_3)$ varies between 62 ppm (for the UK-AV-1 definition) to 78 ppm (for the three values of K_0 within the UK-AV-1 definition (Table 9). This effect of the monitoring and inspection procedure is exceedingly important: it means that the impacts of a standard are highly dependent on the sample size, sampling period, exemption numbers, etc. For example, if EPA had an average standard of 66 combined with their own EPA-AV scheme, such standard would correspond to a UK standard of 62 combined with the UK-AV-1 scheme. "Correspond" here means that the standard would lead to the same decision making and costs to the developer.

Table 9 CUTOFF LEVELS $l_0^*(d_j)$ AT WHICH THE DEVELOPER WOULD SWITCH FROM TREATMENT d_j TO TREATMENT d_{j+1}
(For UK-AV-1 definition)

	$\alpha = 2\cdot10^{-6}$			$\alpha = 0$			$\alpha = -2\cdot10^{-6}$		
$K_0(10^3)$	10	100	1000	10	100	1000	10	100	1000
d_1 (none)	712	749	778	714	754	790	717	757	812
d_2 (G)	140	147	152	140	147	156	140	148	159
d_3 (CPI)	60	62	63	60	62	64	60	62	65
d_4 (CPI+GF)	30	32	33	30	32	34	30	32	35
d_5 (CPI,GF,F)	8.5	8.8	9.2	8.6	9.1	9.8	8.8	9.3	10.5
d_6 (CPI,GF,F,B)	1.3	1.3	1.4	1.3	1.4	1.5	1.4	1.4	1.5
d_7 (reinject)	-	-	-	-	-	-	-	-	-

Table 10 CUTOFF LEVELS $l_0^*(d_j)$ AT WHICH THE DEVELOPER WOULD SWITCH FROM TREATMENT d_j TO TREATMENT d_{j+1}
(For K_0 = 100 000; risk neutral utility function)

	DETECTION DEFINITION*				
	UK-MAX	EPA-AV	NWY-AV	UK-AV-0	UK-AV-1
d_1 (none)	872	767	785	791	754
d_2 (gravity)	172	151	155	153	147
d_3 (CPI)	78	66	64	63	62
d_4 (CPI,GF)	38	34	34	33	32
d_5 (CPI,GF,F)	13.7	10.4	10.1	9.5	9.1
d_6 (CPI,GF,F,B)	2.1	1.6	1.5	1.4	1.4
d_7 (reinject)	-	-	-	-	-

*Precise detection definitions are:

UK-MAX : $t = 1/2$; $n = 1$; $T = 30$, $m = 2$
EPA-AV : $t = 1$; $n = 4$; $T = 30$, $m = 2$
NWY-AV : $t = 1$; $n \to \infty$; $T = 30$, $m = 1$
UK-AV-0 : $t = 30$; $n = 60$; $T = 5400$, $m = 0$
UK-AV-1 : $t = 30$; $n = 60$; $T = 5400$, $m = 1$

The second observation is that the UK-MAX scheme leads the developer to the most cautious cutoff levels, i.e., to a switch to improved treatment, even if the standard is far above the mean $\bar{l}_j$ of the performance uncertainty distribution. On the other extreme, the UK-AV-1 scheme the most lax one since it would allow the developer to stay longer with infereior treatment when the standard level is tightened up.

The final observation from these calculations is that a joint average and maximum standard (as the one which the UK presently uses) can lead the developer into a conflict: should he decide according to the maximum standard or according to the average standard? For example, if the standard was set at sl = 35, the developer model would select treatment d_5 (CPI, Gas Flotation, and Filtering) if it was the maximum standard (UK-MAX), but he would still stick with d_4 (CPI and Gas Flotation) if it was an average standard. Or, alternatively, if the maximum standard is different from the average standard, e.g., $sl_{max} = 100$ and $sl_{av} = 40$, then the developer model would select d_4 according to the average standard, (UK-AV-1), but stick with d_3 for the maximum standard.

To give some illustration of the results in Table 10, and to complete the developer model analysis, Figure 4 shows the optimal developer decisions and the associated expected utilities in the case of risk neutral utilities, for K_0 = 100 000 Pounds, and for the EPA-AV definition. For lax standards say above 900 ppm the developer model chooses no treatment as its optimal decision, and the cost of that decision are small, since the probability of violating such a lax standard is very low (in the above case $p_D(Q_1|900, d_1, sm) = .0000043$ where sm corresponds to the EPA-AV definition). If the standard is tightened up, the probability of detection will increase and therefore the expected utility U_D will decrease. If the cutoff point $l_0^*(d_1)$ is reached, this utility U_D is just equal to the utility of the cost of the next best treatment decision d_2. As can be seen, if the standard is further tightened the utility U_D remains constant, since the probability of detection remains small. Only when reaching stricter levels (e.g., around 160 ppm) this probability increases rapidly and adds to the cost of d_2 the expected penalty cost, therefore further decreasing the utility U_D until the next cutoff point is reached.

The corresponding figure for other definitions of the standard monitoring and inspection procedure will be similar, although the cutoff values will be slightly shifted, and the slope of U_D - function in the area of the cutoff value will change.

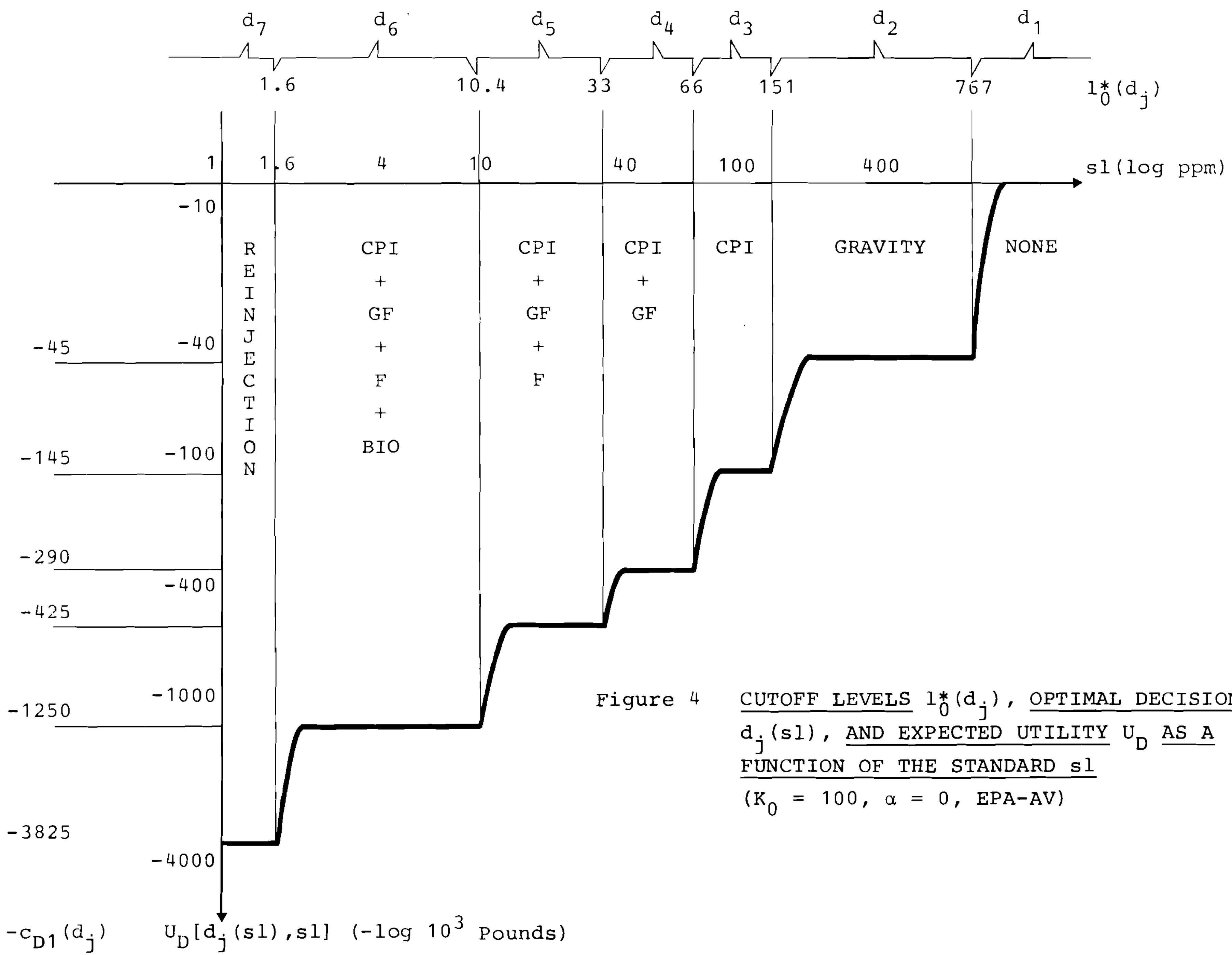

Figure 4 <u>CUTOFF LEVELS</u> $l_0^*(d_j)$, <u>OPTIMAL DECISIONS</u> $d_j(sl)$, <u>AND EXPECTED UTILITY</u> U_D <u>AS A FUNCTION OF THE STANDARD sl</u>
(K_0 = 100, α = 0, EPA-AV)

An interesting way of putting Figure 4 in perspective is by means of a "perfect information analysis". To construct Figure 5 the cutoff levels and expected utilities for the developer's optimal decision making were calculated assuming that there was no equipment performance uncertainty (i.e., $s_j \to 0$ and $\bar{s}_j \to 0$). Overlaying Figures 4 and 5 give some interesting insights. First, now the shift in cutoff levels, before induced through manipulations of K_0 etc., becomes much more drastic. Second, the cost differential around the cutoff levels shows the cost of having to make a decision with imperfect information.

Another change in Figure 4 occurs if another utility function is used. Although Table 9 has already shown that the cutoff levels shift only slightly for different values of α, the utility functions will have effects on the spacing of U_D.

Figures 6 and 7 illustrate this effect for the risk averse and the risk prone decision maker ($\alpha = \pm 2 \cdot 10^{-6}$), K = 100 000 in the EPA-AV scheme. They show clearly that for the risk prone decision maker the relative expected utilities are fairly equally spaced, while for the risk averse decision maker the relative utility of the high cost options are extremely low as compared with the low cost options. This distortion effect even shows although the utilities in Figure 5 are spaced logarithmatically to accomodate the graph.

THE IMPACTEE MODEL

The Problem Structure for the Impactees

This application of the model does not give the impactees any action possibilities. Impactees are considered sufferers of the potential effects of oil pollution. The objective of the impactees are:

1) Minimize mortality of commercial marine organisms (fishery industry);
2) Minimize tainting and chronic toxicity of marine organisms (public);
3) Minimize ecological disturbances (ecologists).

Unfortunately, with the present stage of knowledge about fate and effects of chronic oil pollution, no reasonable estimates can be made about the effects of say oily water emissions of 10 000 tons a day at 100 ppm oil concentration levels. Therefore, it proved futile to attempt to operationalize the three above objectives in terms of value relevant consequence measures (e.g., percent fish loss). In any case, the degree to which the objectives will be met is a function of the random emission levels, and any such function will be highly correlated with actual effects. Rather than operationalizing the above objectives

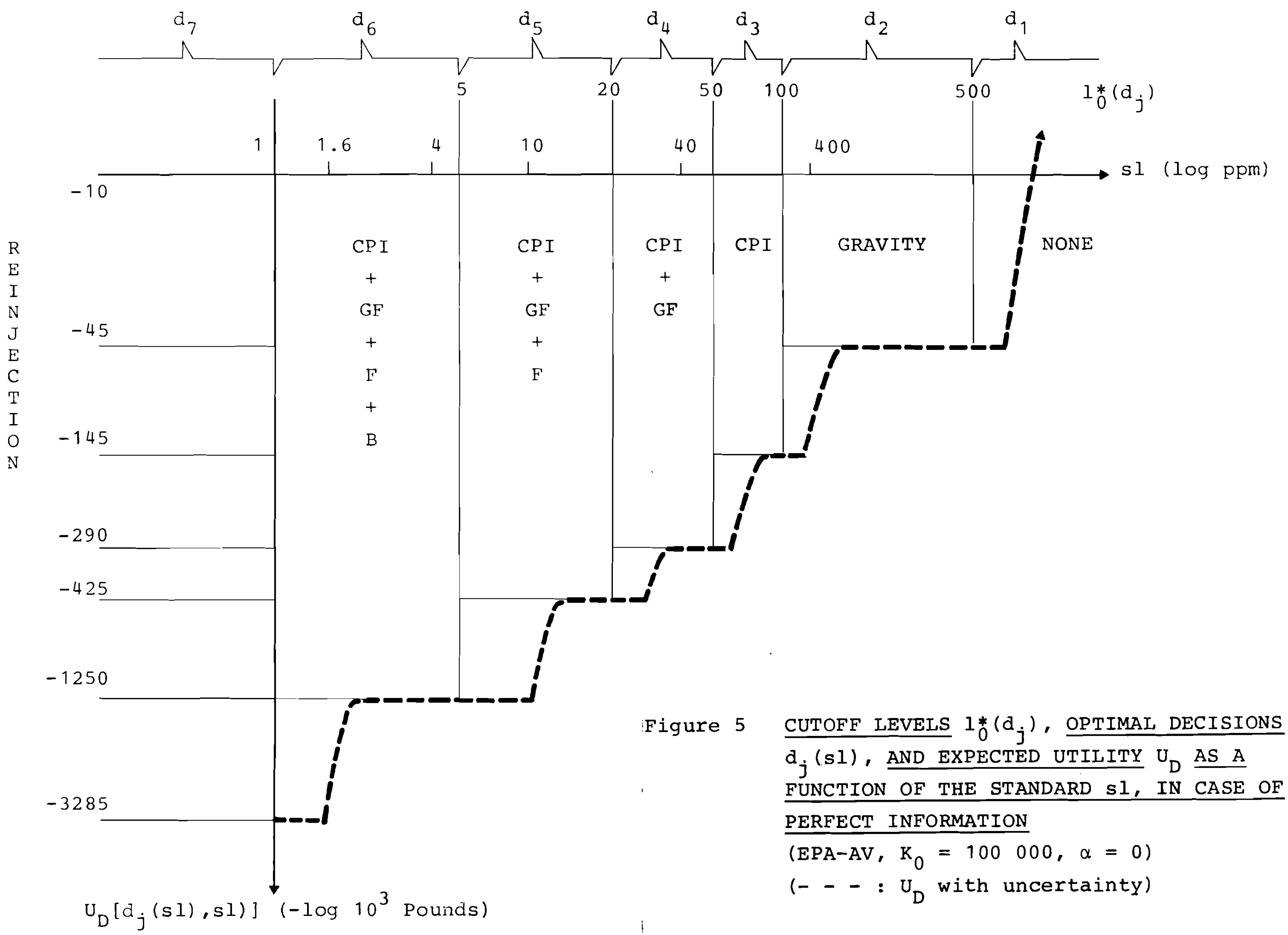

Figure 5 <u>CUTOFF LEVELS</u> $1_0^*(d_j)$, <u>OPTIMAL DECISIONS</u> $d_j(sl)$, <u>AND EXPECTED UTILITY</u> U_D <u>AS A FUNCTION OF THE STANDARD sl, IN CASE OF PERFECT INFORMATION</u>

(EPA-AV, K_0 = 100 000, α = 0)

(- - - : U_D with uncertainty)

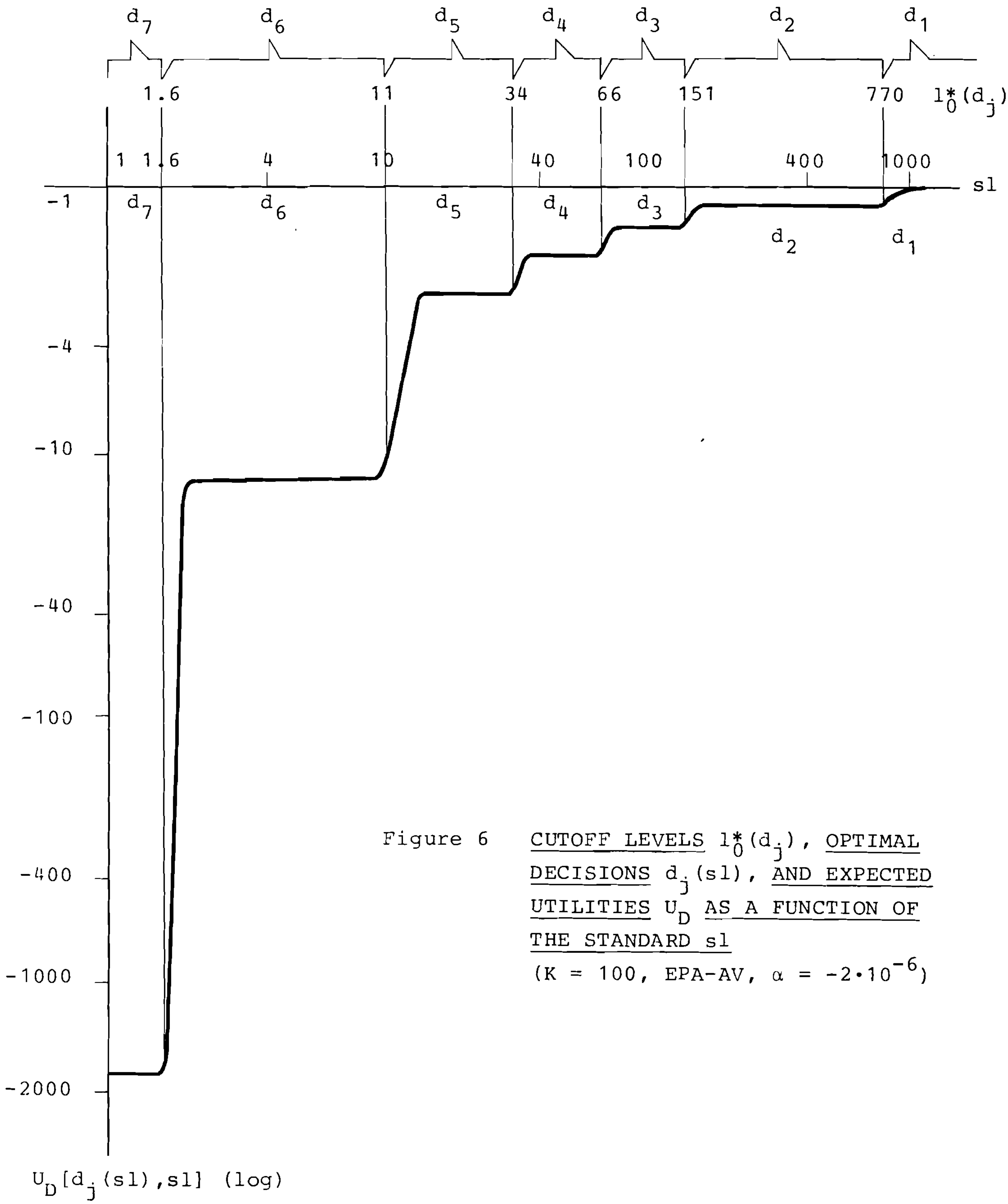

Figure 6 <u>CUTOFF LEVELS</u> $l_0^*(d_j)$, <u>OPTIMAL DECISIONS</u> $d_j(sl)$, <u>AND EXPECTED UTILITIES</u> U_D <u>AS A FUNCTION OF THE STANDARD sl</u>
($K = 100$, EPA-AV, $\alpha = -2 \cdot 10^{-6}$)

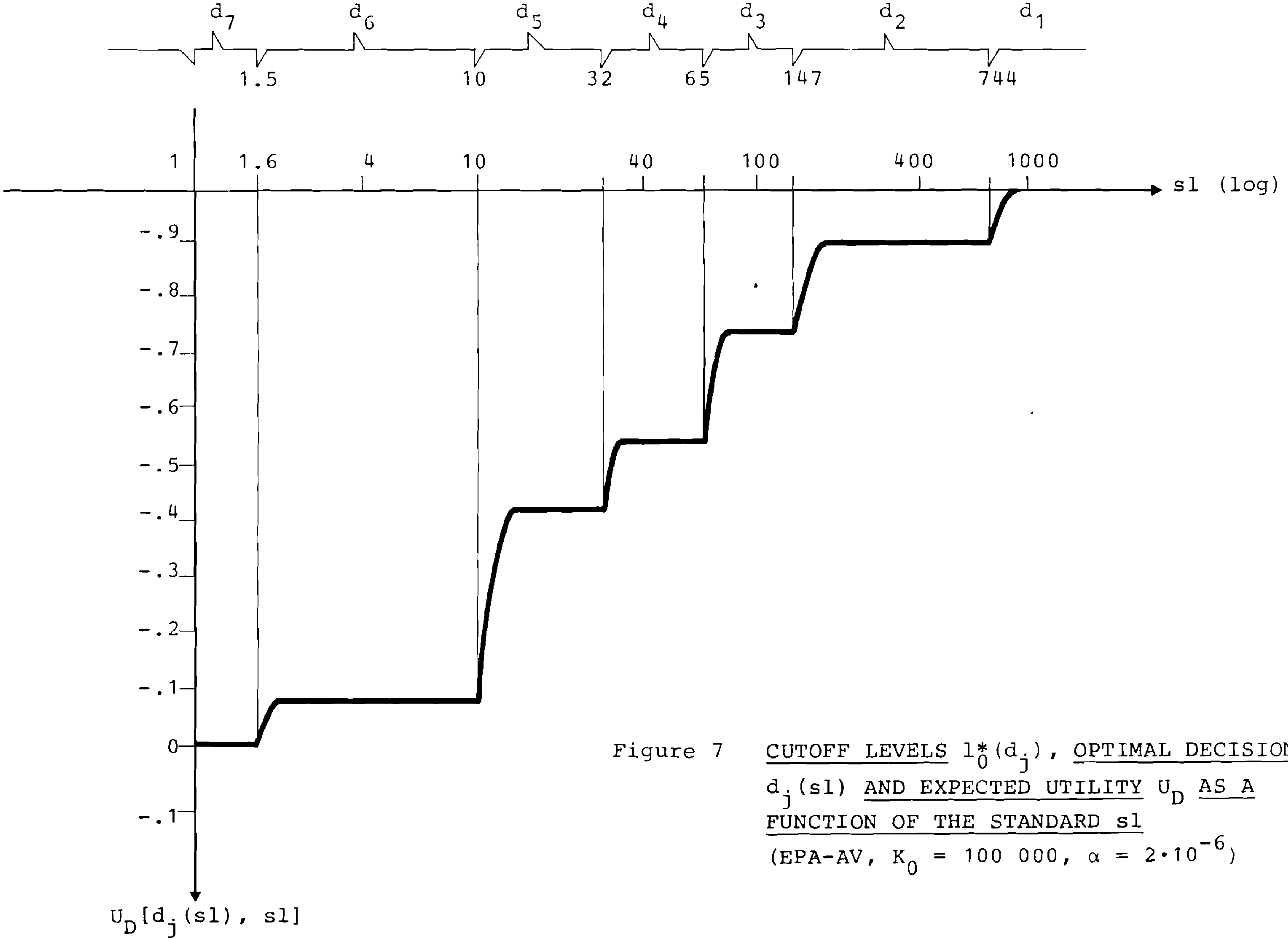

Figure 7 <u>CUTOFF LEVELS</u> $l_0^*(d_j)$, <u>OPTIMAL DECISIONS</u> $d_j(sl)$ <u>AND EXPECTED UTILITY</u> U_D <u>AS A FUNCTION OF THE STANDARD sl</u>

(EPA-AV, $K_0 = 100\ 000$, $\alpha = 2 \cdot 10^{-6}$)

and quantifying uncertainties, as it was done for the developer model, the impactee model will use the random emission levels directly as a proxy for the impactee's consequences. (Using emission levels as a proxy for consequences would, however, be undesirable, if one would want to know more about possible effects of chronic oil pollution and the degrees of uncertainties involved. Also, if one needs to make tradeoffs between costs, pollution and political objectives (i.e., tradoffs between the three decision making units), a further elaboration of the effects of chronic oil pollution would be desirable. However, for the Pareto optimality analysis and the sensitivity analyses to which this paper restricts itself emissions are an appropriate measure.)

Quantification for the Impactee Model

The value function v_A for the impactees is directly defined on the emission level l. This value function is supposed to be linear and negative in l:

$$v_A(l) = -l \tag{49}$$

The probability distributions over emissions l which constitute the events with which the impactees are concerned were already quantified for the developer model. For the impactees the relevant distribution would be the distribution over $\hat{l}$ for n = 1, i.e., the actual diluted performance distribution of the given treatment d_j. It is assumed that the developer and the impactees agree that the distributions given in Table 7 reflect these uncertainties.

As an alternative, one could quantify the value function v_A by an exponential function in order to reflect that increases in emission levels at low initial levels are considered ore or less serious than increases at already high levels. Since an exponential form results if the impactee is assumed to risk seeking, no special attempt will be made to redefine v_A.

According to (6) the impactees' utility function is defined as

$$U_A(d_j) = \int_{-\infty}^{+\infty} N(\bar{l}_j, \hat{s}_j) \cdot (\text{sign } \beta)\ e^{\beta(-l)} dl \tag{50}$$

for $\beta \neq 0$ and as

$$U_A(d_j) = \int_{-\infty}^{+\infty} N(\bar{l}_j, \hat{s}_j) \cdot (-l) dl \tag{51}$$

for $\beta = 0$, $\hat{s}_j$ is determined using n = 1. Besides the risk neutral impact ($\beta = 0$) a risk prone impactee ($\beta = .028$) and a risk averse impactee ($\beta = -.028$) are considered.

Some Model Results for the Impactees

The main analysis for the impactees consists in the computation of the expected utilities $U_A(d_j)$. Table 10 lists the expected utilities for the three utility functions. The expected utilities for the exponential utility functions could be determined by observing that the moment generating function of the normal distribution is

$$M(t) = E(e^{tx}) = e^{tm + \frac{t^2 s^2}{2}} \tag{53}$$

where m and s are the mean and the standard deviation of the normal distribution. Thus setting t = .028 and -.028 the expected utilities could be calculated from the means and the standard deviations of the normal distributions for the d_j's. For the linear utility function the expected utilities coincide with the means $\bar{l}_j$. Table 11 summarizes the results.

The effect that already showed for the developer model, namely that expected utilities for the risk prone decision maker are much more equally spaced than those for the risk seeking one shows here again.

DOMINANCE ANALYSIS

The emphasis of the modelling attempt so far was put mainly on the developer model through which it was possible to determine the developer's response $d_j(sl)$ as a function of a standard sl, for various monitoring and inspection procedures sm, penalty schemes and utility functions. In comparison to the detailed treatment of the developer model, both the regulator model and the impactee model are relatively superficial. This reflects partly the better data base of the developer model, partly the softness of modelling the "political" decision making of the regulator, and partly the high uncertainty of the impactee model which prohibited a detailed analysis of fate and effects of oil pollution. The following sections will nevertheless try to convey a coherent picture of the accomplished results of all three decisions models. First the sequence r, d(r), $U_R(r)$, $U_D[d(r),r]$, and $U_A[d(r)]$ will be examined for various cases as outlined in the general scheme of the regulator-developer-impactee model of p. 7. Comparing utilities U_R, U_D, and U_A then leads naturally into a Pareto optimality analysis which determines ordinally dominated standards, i.e., standards which are not better (in terms of utility) for any decision maker, and worse for at least one. It will be shown that such standards actually exist due to the discreteness of the developer's response.

Table 11 EXPECTED UTILITIES $U_A(d_j)$ FOR IMPACTEES FOR THREE DIFFERENT UTILITY FUNCTIONS

(Utility Functions are Standardized such that $U_A(d_2) = 0$; $U_A(d_7) = 100$)

TREATMENT	MEAN EMISSION LEVEL $\bar{l}_j$	RISK PRONE $\beta = -.028$	RISK NEUTRAL $\beta = 0$	RISK AVERSE $\beta = +.028$
d_1 (none)	500	-12.6	-400.0	$-1.3 \cdot 10^8$
d_2 (gravity)	100	0.0	0.0	0.0
d_3 (CPI)	50	17.9	50.0	84.8
d_4 (CPI,GF)	20	43.7	80.0	95.1
d_5 (CPI,GF,F)	5	86.3	95.0	99.2
d_6 (GPI,GF,F,B)	1	97.0	99.0	99.8
d_7 (reinject)	0	100.0	100.0	100.0

Table 12 gives an example, in which the EPA-AV monitoring and inspection scheme was used, for the risk neutral developer and impactee utility function, for the unit weighting scheme of the regulator, and a penalty value of K_0 = 100 000. For the purpose of constructing this table the utilities of all three decision making groups were standardized to cover similar ranges. The most interesting result is that there are many dominated standards, i.e., standards which are not better than others for anyone, but worse for at least one. For example, the standard of 50 ppm results in utilities of 48 to the regulator, 28 for the developer, and 80 for the impactee. However, the standard of 35 ppm results in the same decision of the developer, and thus the same utilities for the developer and the impactee as the standard of 50 ppm, while the regulator's utility increases to 65. In this case the standard of 50 ppm would be considered dominated by the standard of 35 (and, in fact, by any standard between 35 and 50). The model would suggest to eliminate this standard from further consideration.

Table 12 UTILITIES OF REGULATOR, DEVELOPER, AND IMPACTEES AS A FUNCTION OF THE STANDARD sl

(α, β = 0, K_0 = 100 000, EPA-AV scheme, unit weights w_{Ri})

sl	$d_j(sl)$	$U_R(sl)$	$U_D[d_j(sl)]$	$U_A(d_j)$
0	d_7	26	-856	100
1	d_7	26.5	-856	100
2	d_6	27	-213	99
5	d_6	29	-213	99
10	d_6	30	-213	99
15	d_5	38	-6	95
20	d_5	48	-6	95
35	d_4	65	28	80
40	d_4	62	28	80
50	d_4	48	28	80
100	d_3	28	64	50
150	d_3	26	64	50
500	d_2	25	89	0
1000	d_1	25	100	-400

The same effect is shown in Table 13 for the analysis of the UK-MAX monitoring and inspection procedure, where the only other change is the selection of the differential weighting scheme for the regulator's utility function. Again the dominance effect shows. Consider, for example, the maximum standard of 150 ppm vs. the standard of 100 ppm. In both cases the same utilities accrue to the developer and the impactee. However, the regulator would prefer 150 ppm. Therefore 150 ppm dominates 100 ppm.

The reason for these dominance effects lies in the formulation of the decision problem. In the model the regulator's utility function changes continuously with the standard sl. However, the developer's responses to standards are discrete, with segments of standard levels in which the developer would select the same treatment response. Within most of the range between two adjacent cutoff points l_0^* the utility of the developer does not change* and begins to decrease only if the standard approaches the next cutoff point. Only when the standard is very close to this cutoff point (e.g., 1-5 ppm) a rapid decrease in utility occurs. If the standard is further away the probability of detection is extremely small so that there is virtually no change in utility. The impactees' utility depends only on the developer's action. They stay constant in the full range of standards between two adjacent cutoff points l_0^*. Therefore within most of the range between two such cutoff points the regulator's utility dominate.

Thus a fine analysis of the dominance phenomenon has to focus on the small range around the cutoff points in which there is a noticeable change in the developer's utility function. Since between any two adjacent cutoff points the impactees' utility remains constant, the analysis can be done by plotting the regulator's vs. the developer's utilities. Figure 8 shows such a plot for the EPA-AV scheme, K_0 = 100 000, risk neutral utility functions, and the unit weighting scheme for the regulator's utility function. The figure shows the changes in utility between a standard of 33 and the standard of 767.

There are several remarkable features in this figure. First of all the figure demonstrates again how fast the developer's utility function changes between the cutoff points. Consider the cutoff point of 33 ppm. Between 33 and 36 ppm the developer's utility function changes from -6 to +28 (this corresponds to a monetary change from 425 000 to 290 000 Pounds). For such a small change in the standard, the regulator's utility function changes, of course, only very little. Then, after 36 ppm the regulator's utility function changes, but up to 66 ppm the developer's utility function remains virtually constant. A similar picture arises then after the next cutoff point of 66 ppm.

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* Strictly speaking this is not true since there are, of course, changes in the developer's utility function even at values only slightly lower than the cutoff value. However, for all practical purposes, the probability of detection is so small that these changes can be neglected.

Table 13 UTILITIES FOR REGULATOR, DEVELOPER, AND IMPACTEES AS A FUNCTION OF THE STANDARD sl

(UK-MAX scheme, $\alpha, \beta = 0$, $K_0 = 100\ 000$, diff. weights w_{Ri})

sl	$d_j(sl)$	$U_R(sl)$	$U_D[d_j(sl)]$	$U_A(d_j)$
0	d_7	22	-856	100
1	d_7	22	-856	100
2	d_7	23	-856	100
5	d_6	25	-213	99
10	d_6	29	-213	99
15	d_5	36	-6	95
20	d_5	45	-6	95
35	d_5	65	-6	95
40	d_4	64	28	80
50	d_4	56	28	80
100	d_3	36	64	50
150	d_3	38	64	50
500	d_2	40	89	0
1000	d_1	40	100	-400

Therefore, although there is a standard for each point on the line in Figure 8, some of them can be eliminated through dominance. For such analysis, however, also the changes in the impactees' utilities have to be considered. Figure 9 shows the developer vs. impactees' utilities for the standards in Figure 8. There is no change in the impactees' utility function between standards of 11 and 32 ppm. At 33 ppm the utility function drops to 80 and stays constant up to the next cutoff level of 66. Of course the developer's utility function shows an increase in that range. At 66 ppm the impactees' utility function drops again due to the changed decision by the developer. Again, until the next cutoff level of 151, no change occurs in the impactees' utilities while the developer improves his utilities.

Which standard would survive a dominance test? In strict mathematical terms there is no dominated standard, since the developer's utility function changes, however slightly, also for standard values below a cutoff point. From an inspection of Figures 8 and 9 however, it is clear that standards in the range between 40 and 66 ppm, between 70 and 151 ppm, and 160 and 767 ppm produce essentially no change in the developer's utility function. The question then is, how close to the cutoff point must the standard be in order to consider a utility change

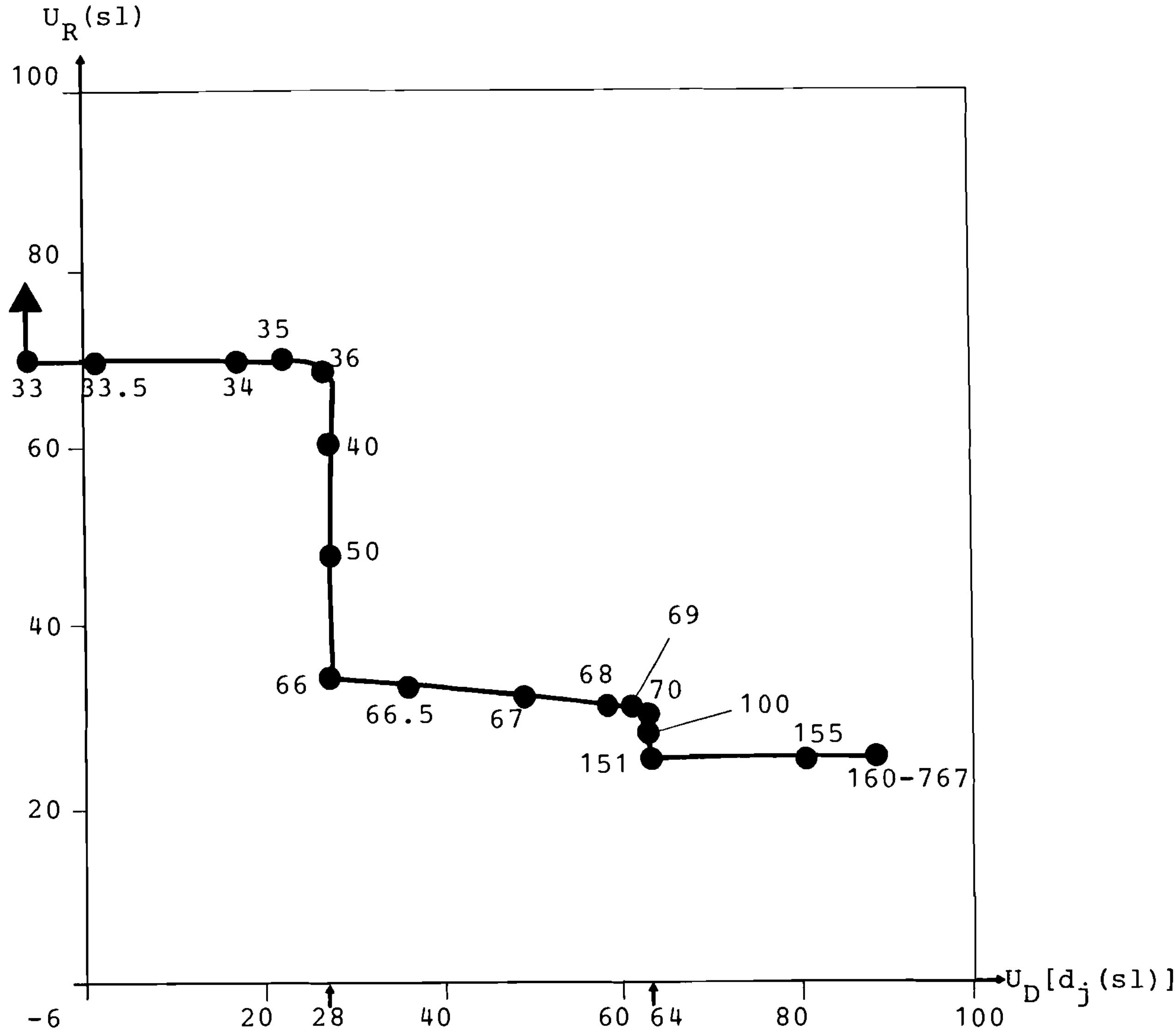

Figure 8 <u>STANDARDS BETWEEN 33 AND 767 IN THE REGULATOR VS. DEVELOPER UTILITY PLANE</u>

(EPA-AV, α = 0, K_0 = 100 000, unit weights w_{Ri})

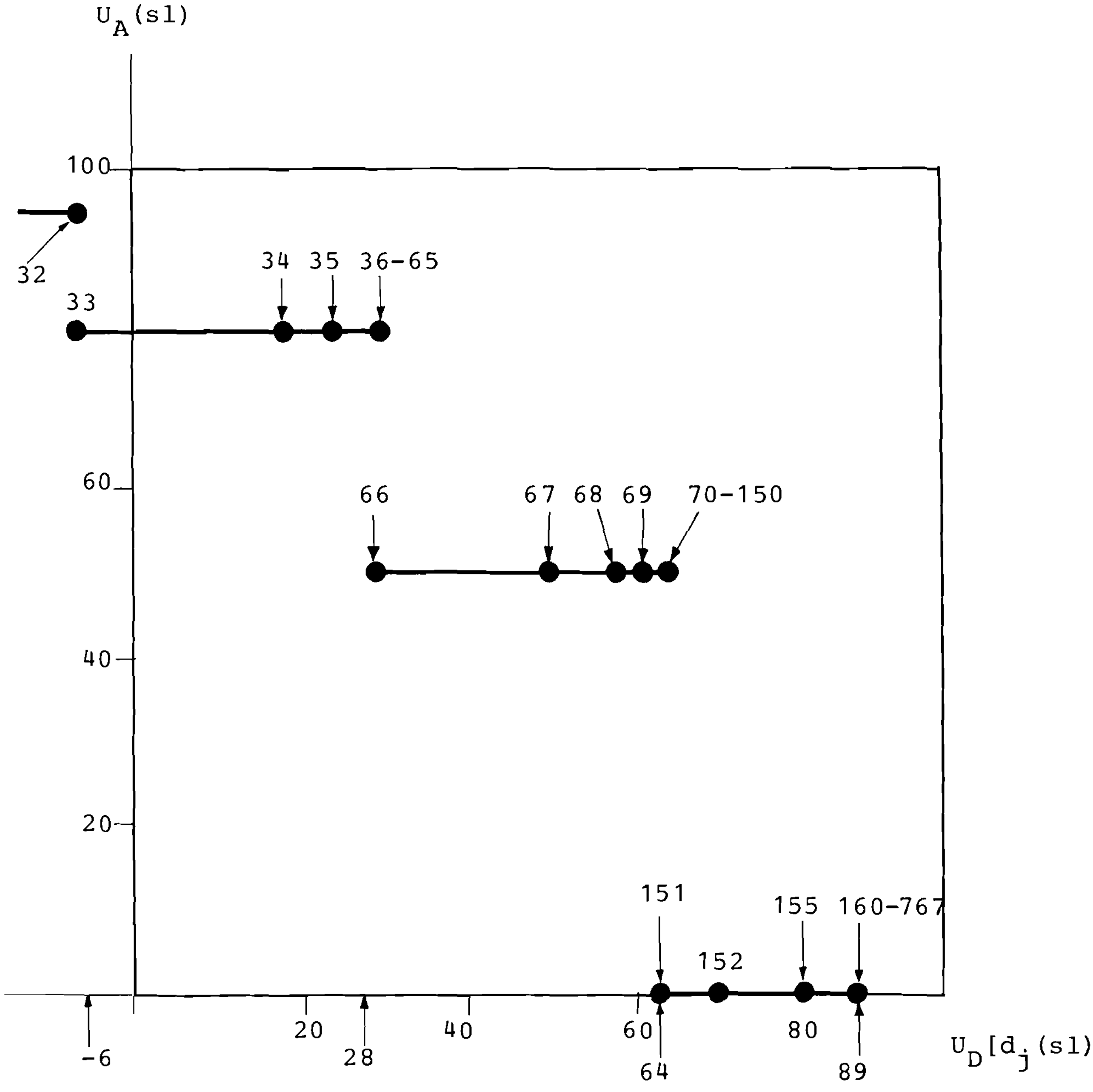

Figure 9 STANDARDS BETWEEN 33 AND 767 IN THE DEVELOPER-IMPACTEE UTILITY PLANE
(EPA-AV, K_0 = 100 000, risk neutral utilities)

significant. Between that point and the next higher cutoff level one could then consider the developer's utility constant, and since the impactee's utility is constant in that range anyway, the regulator's utility would dominate.

For the purpose of such an approximate dominance analysis only integer valued standards are considered and a deviation of the utility function corresponding to less than 100 Pounds is considered not essential. (In the utility function scaling of Figures 8 and 9 one utility unit corresponds to 4027 Pounds, 100 Pounds therefore corresponds to 0.025 units.)

Standards between 37 and 66, between 71 and 151, and between 169 and 767 produce developer's utilities which differ by less than .025 units. These standards are therefore considered dominated by the regulator's utility, which drops with the standard level. The standards which are considered non-dominated are those between the lowest of these dominated standards (e.g., 37) and the next lowest cutoff point (e.g., 33). In this small range the developer's utility function decreases rapidly, while the regulator's utility function improves slightly. Figure 10 shows the standards which would survive this dominance test. All standards between 152 and 168 survive, since the utility change of the regulator is significant according to the above rules.

The results of the dominance analysis depend critically on the cutoff point values l_0^* and the variance of the equipment performance distribution. If one considers the case of perfect information ($\hat{s}_j \rightarrow 0$) detection probabilities are degenerate and therefore no gradual changes of the developer's utility function occur. Instead, the developer's utility function changes in steps as indicated in Figure 5. The points at which the utility changes are the cutoff points for perfect information which coincide with the mean values $\bar{l}_j$ which now have a probability of 1 to occur with treatment d_j. The developer would select d_j for $\bar{l}_j \leq sl < \bar{l}_{j-1}$, e.g., he would choose d_4 (CPI and GF) for $50 \leq sl < 100$. Within that range his utility function and the function of the impactees remain constant, and the regulator's utility function dominates. Figure 11 shows the resulting non-dominated standards for this perfect information case.

50, 100, and 500 ppm are cutoff points and at the same time the points with the highest regulator's utility in the respective intervals [50, 100); [100, 500); and [500, ∞). 32 is the point with the highest regulator utility in the interval [20, 50), and 19 is the integer valued standard with the highest regulator utility in the interval [5, 20). (Although 19 is dominated in the regulator-developer utility plane, it is not dominated in the developer-impactee plane).

The perfect information analysis shows clearly the dependence of non-dominated standards on the degree of uncertainty (as expressed in $\hat{s}_j$). An interesting byproduct of the perfect

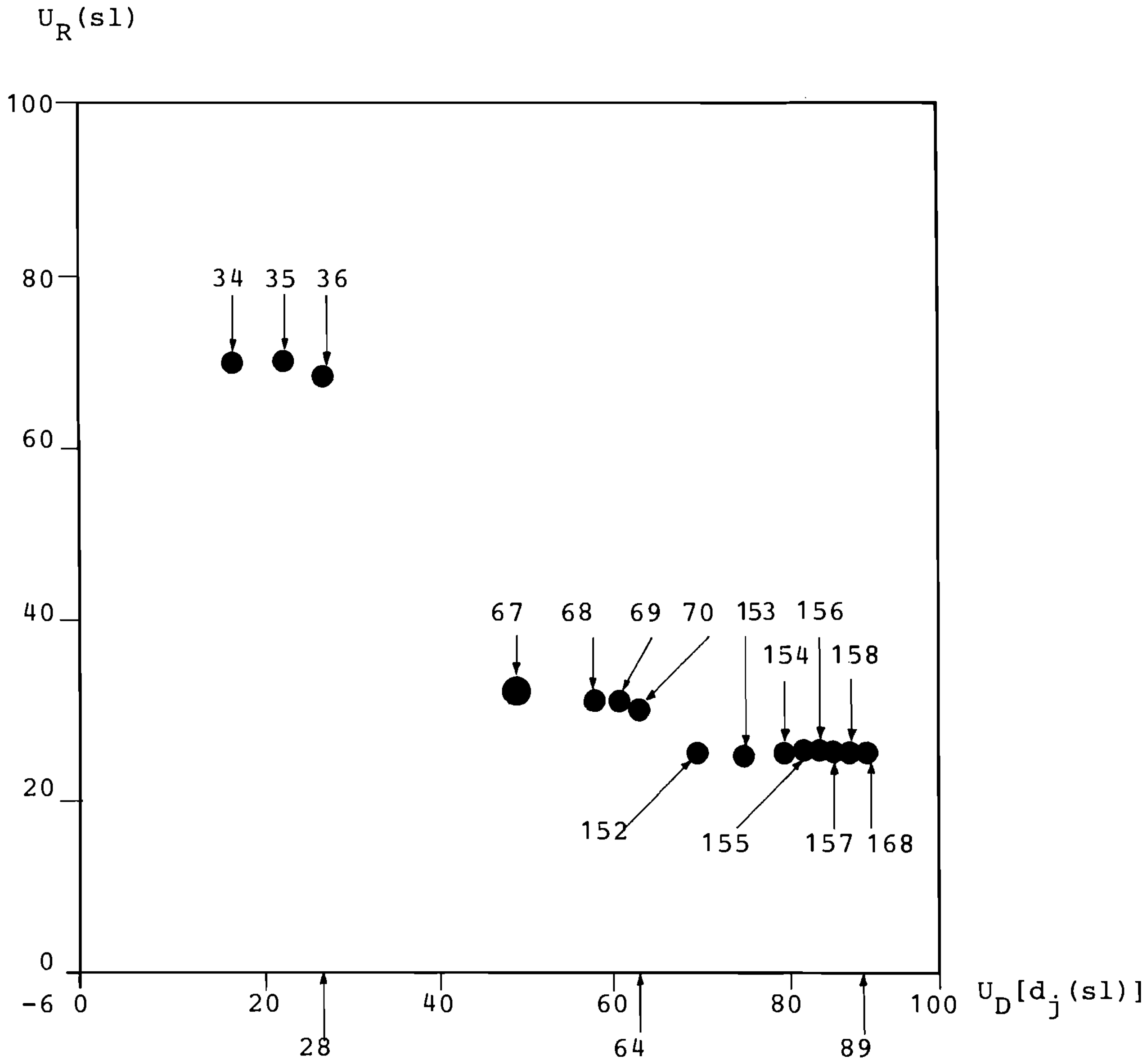

Figure 10 NON-DOMINATED STANDARDS BETWEEN 33 AND 767 IN THE REGULATOR-DEVELOPER UTILITY PLANE
(EPA-AV, K_0 = 100 000, risk neutral utilities, unit weights w_{Ri})

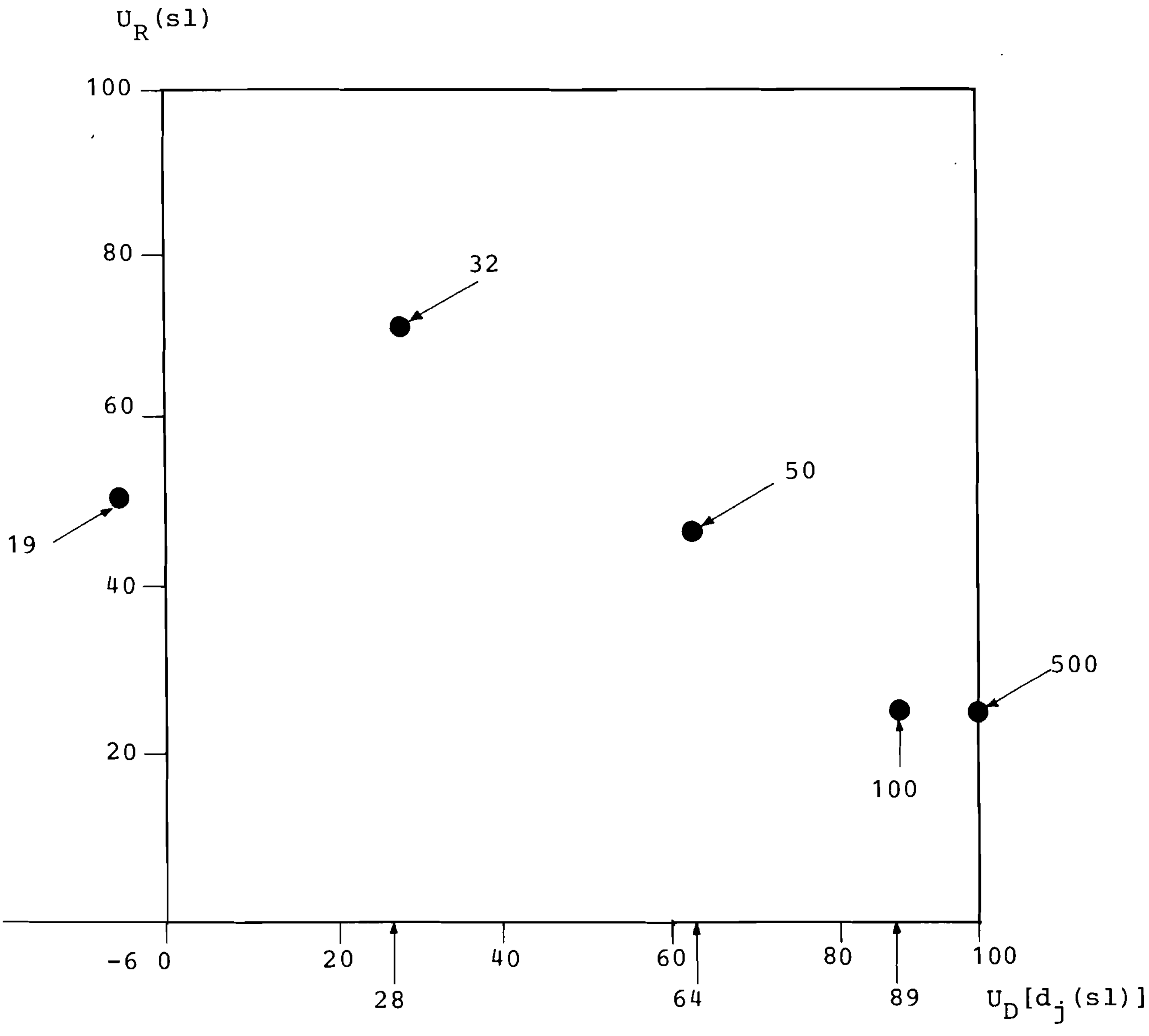

Figure 11 NON-DOMINATED STANDARDS BETWEEN 5 AND 500 PPM IN THE REGULATOR-DEVELOPER UTILITY PLANE UNDER PERFECT INFORMATION

(EPA-AV, K_0 = 100 000, risk neutral utilities, unit weights w_{Ri})

information analysis is that 32 emerges as a non-dominated standard here, while 34-36 were non-dominated in the previous analysis. The reasons are, however quite different: in the perfect information analysis 32 dominates because it coincides with the peak of the regulator's utility function, while in the previous analysis 34-36 dominate because they are just above the next lowest cutoff point $l_0^*(d_4) = 33$.

In inspecting Figures 8 to 11 it becomes clear that in the "normal" range of possible standards, e.g., between 10 and 70 ppm, certain groupings of standards are more desirable than others. These are groupings of standards around cutoff points or around the maximum of the regulator's utility function. Depending on uncertainty, the cutoff points will change. It appears, however, that for several uncertainty characteristics and several changes in model parameters standard values between 30 and 40 ppm emerge as "good" (in a non-technical sense) solutions; they are "good" in the regulator's sense since they lie in a range around the maximum of the regulator's utility function. They are acceptable for the developer, as long as they are not too strict as to force him to select d_5; and finally, they are acceptable to the impactees since their utilities reach 80 for such standards (as compared to 0 for 500 ppm and 100 for 0 ppm).

DISCUSSION

From running a decision theoretic model with many soft, several uncertain, and some even speculative quantitative inputs, no final decision recommendations can be expected. In fact, recommendation was not the primary purpose of running the model but rather a feasibility test of the model itself (does it behave right?) and a sensitivity and dominance analysis of the parts of the chronic oil regulation problem which are of most interest to the regulator, the developer, and the impactees.

In the following first some general remarks about the applicability and possible problems of applying the decision theoretic model to standard setting will be made, followed by a discussion of the interesting model sensitivities, dominance effects, and implications for real life standard setting.

In principle the model seems applicable in the sense that it is understandable to the experts and representatives of the decision units involved, that it reflects the basic structure of the decision problem, and that it behaves "right" (as one would want a decision model to behave), both in its overall characteristics (dominance effects, changes of utility function), as well as in its local behavior (changes of cutoff points, sensitivity to uncertainty, etc.).

The model is weakest in the regulator and the impactee part. One may want to consider better procedures to establish and quantify the political objectives of the regulator, for example, through scenario approaches. The impactee part can certainly be improved in other applications. The fact that impactees did not have response capability in the model and that they were assumed to follow such a simple objective function is not only an oversimplification but probably an undesirable way to go about modelling impactees. In most standard setting problems one would need a more formal way to deal with the fates and effects of pollutants, and decision theory certainly offers tools to do just that. In this illustrative application, no such attempt was made partly because marine biologists have great difficulties in answering even the most basic questions about chronic oil pollution.

Improvements could certainly be done here by using the ambient distribution as an index C_A to measure the impactees' objectives. Diffusion models for continuous point source emissions into an infinite disk would be appropriate and were, in fact explored. However, the uncertainty about the model parameters (decay rate, diffusion coefficients) for North Sea conditions ultimately would have made the application of a more sophisticated model less useful than using emissions directly.

The developer model emerges as the heart of the present modelling attempt, and in fact, most of what can be learned from this model application is a result of the developer model run. It is clear that the developer model had an advantage by being easier to quantify (uncertainties about equipment, cost, etc.), through the inherent statistical definitions of standards, and equipment performance, and the relatively good data base. Yet, here too some improvements should be made in real applications.

The no-discounting of future operation and sanction cost is certainly unrealistic. If discounting is done, however, the model becomes quite a bit more complicated, since now the exact time of detection matters. The model assumption of zero probability of detection after a treatment improvement is unrealistic, but with the present quantification proved reasonable. In real applications finer choices of the developer may be included which then would leave the possibility of detection after improvement. All these ideas for improvement really lead up to a more finely tuned decision making model, made explicitely sequential, taking into account the continuous inflow of information, etc.

For the developer model alone such improvement appears possible, since the structure of the sequential problem is rather well defined and analysed in the literature (see, for example, DeGroot, 1970; Rapaport, 1967). However dynamizing the model in its present three decision maker version would lead to serious problems, since it is now lacking feedback loops from the developer and impactee back to the regulator etc. In the present one stage formulation this is still acceptable, since the model is considered more a cognotive roadmap for the interest parties along which they can test short term reactions and responses, rather than a simulation of a real sequential and interactive decision process. For a step in the direction of sequential multi-stage games see Höpfinger and Avenhaus, 1978.

So much for a discussion of the general model characteristics, problems, and possibilities which this illustrative application encountered. This discussion will conclude with some more specific remarks about the sensitivity and dominance results which the model uncovered. This discussion will largely be restricted to the detection and the developer model.

The model clearly showed the influence of the uncertainty about equipment performance on the developer's decision, i.e., the definition of cutoff levels l_0^*. Since this uncertainty is also influenced by the monitoring and inspection procedure, the model demonstrated that the definitions of such procedures and the definitions of detection probabilities can play a crucial part in standard setting.

For most penalty schemes, utility functions, and other parameters, the cutoff levels l_0^* were approximately 2-3 standard deviations higher than the corresponding mean of the equipment performance distribution $\bar{l}_j$. This insensitivity has a very simple explanation in the model: detection probabilities over the lifetime of the plant (as defined e.g., by EPA-AV or UK-MAX) which are in the range of .01 to .99 are all generated by single day violation probabilities ($\hat{l} > sl$) of .02 to .0001. But probabilities in that range all correspond to areas under the normal distribution outside of 2-3 standard deviations from the mean. Since cutoff levels l_0^* were directly determined from cutoff probabilities p_D^* between .01 and .99 this insensitivity becomes understandable.

In the modelling of detection probabilities another result of the model run emerged: a standard is not equal to a standard. Since sample size, sampling period, inspection interval, and exemption number all influence the uncertainty about $\hat{l}$ and detections, a tighter numerical standard can be in fact more lax than a numerically higher one. Demonstrations of this effect were given in the discussion of the developer model.

The most interesting result in the dominance analysis came from the discreteness of the developer's response to the continuous decision of the regulator. Together with the "suffering" role of the impactees, this paradigm allowed the regulator to dominate decisions with his utility function.

Not all of these model features may apply to other standard setting cases. Some may not even remain tenable in a more detailed run of the model for chronic oil discharges. A few main conclusions can however already be drawn:

1) It is very important how the regulator defines sampling, monitoring procedures and detection states;

2) There will be in many cases dominated standards, and the regulator should find the sensitive non-dominated areas;

3) Uncertainty about equipment performance is a <u>crucial</u> parameter to determine the developer's response to a standard; and

4) Penalties do not control the developer's response to a strong degree.

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DYNAMIC STANDARD SETTING FOR CARBON DIOXIDE*

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Dynamic Standard Setting for Carbon Dioxide

by

E. Höpfinger

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Abstract

Under the assumption that a continuous increase in atmospheric carbon dioxide beyond a critical value, caused by the combustion of fossil fuel, will lead to irreversible and large changes of the climate of the earth, the problem of limiting CO_2 emission becomes an urgent concern. The subject of how to determine and adapt an emission standard for carbon dioxide is treated as a three-person infinite stage game, the players of which are the decision units of regulators, producers, and population. After the description of the model solutions are derived for several solution concepts and discussed. In special cases the solutions differ substantially from each other.

Dynamic Standard Setting for Carbon Dioxide

INTRODUCTION

The emission of carbon dioxide into the atmosphere resulting from fossil fuel use has been increasing at an exponential rate for more than one century. If this expansion continues, the concentration of carbon dioxide in the atmosphere may be doubled in about the next 60 years according to R.M. Rotty, 1977. The effects on the global climate may well appear suddenly and could get out of control before remedial actions become effective.

Since easily accessible fossil fuels contain such big amounts of carbon there is a strong tendency to use them as a source of energy that could last for nearly two more centuries. This is much more so since the competing nuclear energy meets increasing resistance by citizen groups. But it is the vastness of this carbon reserve that causes deep concern within the climatological community. The amount of carbon in recoverable fossil reserves is ten times the amount now contained as carbon dioxide in the entire global atmosphere.

As these reserves are being used, the concentration of carbon dioxide in the atmosphere will surely increase; and because carbon dioxide absorbs a portion of the infrared radiation emitted by the earth, it is generally believed that a higher atmospheric temperature will result ("greenhouse effect"). Although it is uncertain how much warming is produced by a given increase, the increased atmospheric carbon dioxide could have a considerable impact on man's environment.

Significant physical effects that may be expected with high fossil use are the melting of polar sea ice and/or decreasing precipitation in mid-latitude regions. Major socio-political impacts could plausibly attend a substantial increase of carbon dioxide, for example:

- large and persistent fluctuations in global food supply, due to repeated crop failures in various regions of the world which are caused by chronic and severe weather variability;

- increasingly regulated demographic migration between regions and across national borders, due to a climate-related collapse of selected webs in regional economies; shifts in the power balance among nations due to physical effects stimulating the economic and cultural decline in some regions and stimulating increased growth and prosperity elsewhere.

At the present time the physical processes causing variations of temperature are poorly understood (see J. Williams, 1978 and T. Augustsson et al., 1977), and changes due to atmospheric carbon dioxide increases are impossible to detect since there is no accurate knowledge of the natural variability of the global average temperature. As outlined by O.W. Markley et al., 1977, and R.M. Rotty, 1977, the other physical and sociopolitical effects are also highly uncertain.

Although a large part of the climatological community shares the opinion that mankind needs and can afford a time window between five and ten years for vigorous research and planning in order to narrow the uncertainties sufficiently so as to justify a major change in energy policies, the model analyzed in this paper excludes an increase of relevant knowledge about the physical effects. Thus the model deals with the pessimistic view of the climatic aspects of carbon dioxide. It is global in character because the global effects seem to dominate the local or regional ones.

Given these substantial uncertainties about the development of climate, the problem of what energy policies governments should choose, becomes important. This problem is approached as a conflict situation among the groups of governments, producers emitting carbon dioxide, and population. In order to work out the global aspects this conflict situation has been formalized as a multistage three person game, the players of which are called regulator, producer, and impactee. Thus we neglect conflicting interests among governments, producers, and different groups of populations, such as of developed and developing countries. The regulator stands for an international agency, the producer for an organization of all producers, and the impactee for the community of people possibly affected by the carbon dioxide problem.

The paper is based on the assumption that a continuous increase of atmospheric carbon dioxide beyond a critical value will lead to irreversible and large changes of the climate which are regarded as a catastrophe. All three players have their subjective probability of the level of the critical value. Since, by assumption, there is no increase of knowledge about the climatological process, the regulator can only be concerned about the reactions of the producer and especially of the impactee.

After the specification of the model the results for several solution concepts are derived. These are quite different in general but can all be interpreted in terms of fair play or power. Given that the model allows prescriptive answers although it is primarily descriptive.

Since data are often unknown or scarcely available or arbitrary--as in the case of the regulator where the utility function may be conceived of as reflecting a trade-off between the interests of producer and impactee--solutions are derived as functions of the parameters. Hence parameter analysis can reveal the crucial parameters. For the purpose of illustration a small numerical example is added.

THE MODEL

The conflict situation is described by a three-person dynamic or multistage game in extensive form (see G. Owen, 1968, or J.C.C. McKinsey, 1952) which resembles stochastic games. At each stage a component game of perfect information is played which is completely specified by a state. The players' choices control not only the payoffs but also the transition probabilities governing the game to be played at the next stage. Each player has his own subjective estimate of the transition probability due to his subjective probability of the "true critical value".

The set of states of the game is

$$S = \{(C,L) \mid C_p \geqslant C \geqslant 0, L \geqslant 0\} \cup \{k \geqslant 0\}$$

C being the amount of carbon dioxide in the atmosphere;
C_p the maximal amount of carbon dioxide if all fossil fuel is burnt;
L the upper bound of carbon dioxide emission during a period;
k the critical value for a catastrophe.

Let (C^1,L^1) denote the first state. Then C^1 can be assigned the present amount of atmospheric carbon dioxide, and L^1 the present maximal emission of CO_2 or some multiple of it.

The perfect information of the component games is specified as follows:

For state (C,L) the regulator's set of choices is

$$M_R(C,L) = \{\ell \mid 0 \leqslant \ell \leqslant L \leqslant L\} \quad ,$$

where ℓ denotes the upper bound of the emission of carbon dioxide by the producer.

Then the producer chooses the amount of carbon dioxide to be emitted. His set of choices or measures equals

$$M_p(C,L,\ell) = \{a \mid 0 \leqslant a \leqslant \ell,\ a \leqslant \frac{C_p - C}{\beta}\} \quad ;$$

$0 < \beta < 1$ is defined below. The impactee's set of measures equals

$$M_I(C,L,\ell,a) = \{p \mid 0 \leqslant p \leqslant 1\} \quad .$$

Knowing the choices ℓ and a he chooses the degree p of the pressure he wants to exert on the regulator. p can denote the probability of a vote to suspend the government or of an aggression against institutions.

The sets of measures in the case of k, i.e. a catastrophe has occureed at amount k of carbon dioxide in the atmosphere, equal

$$M_R(k) = \{0\} \quad ;$$

$$M_P(k,0) = \{0\} \quad ;$$

$$M_I(k,0,0) = \{0\} \quad ;$$

which means that there is no pressure.

Given state (C,L) and the choices (ℓ,a,p) the following states are possible at the next stage:

$$(C + \beta a,\ L), \qquad (C + \beta a,\ \tfrac{L}{2}), \qquad \{k \geqslant C\} \quad .$$

The first component of the first and second states indicates that the constant share βa of emitted carbon dioxide is added to the amount of carbon dioxide in the atmosphere. This is consistent with results of box models for the CO_2 cycle of the earth (see R. Avenhaus, et al., 1978) if a is emitted at a constant rate during the time period. The estimates for β range between 0.01 and 0.5. Amount $(1 - \beta)a$ is assumed to disappear into the biosphere, the upper mixed layer of the sea, and the deep sea. The second components express that the old upper bound either remains or is reduced by half. It is assumed that there is a probability pv that L is replaced by $\frac{L}{2}$, where $0 < v < 1$ is a parameter provided that the catastrophe will not occur. $k \geqslant C$ denotes the amoung of carbon dioxide in the atmosphere at which the catastrophe occurs.

All three players are assumed to have subjective probabilities relating to the critical amount k of carbon dioxide. They characterize the transition probabilities. For simplification of the model we assume that the subjective probabilities concentrate on points denoted by C_R, C_P, and C_I for regulator, producer, and impactee. We assume $C_R < C_P$, $C_I < C_P$ thus allowing the producer to neglect a possible catastrophe.

The subjective probabilities P_R, P_P, P_I for the transition from (C,L) to the possible new states are

New state t	$P_R(t\|C,L,\ell,a,p)$	$P_P(t\|C,L,\ell,a,p)$	$P_I(t\|C,L,\ell,a,p)$
$(C+\beta a, L)$	0 if $C\leqslant C_R<C+\beta a$ or $C_R<C<C+\beta a$ 1-pv if $C+\beta a\leqslant C_R$ or $C_R<C=C+\beta a$	1-pv	0 if $C\leqslant C_I<C+\beta a$ or $C_I<C<C+\beta a$ 1-pv if $C+\beta a\leqslant C_I$ or $C_I<C+\beta a$
$(C+\beta a, \frac{L}{2})$	0 if $C\leqslant C_R<C+\beta a$ or $C_R<C<C+\beta a$ pv if $C+\beta a\leqslant C_R$ or $C_R<C=C+\beta a$	pv	0 if $C\leqslant C_I<C+\beta a$ or $C_I<C<C+\beta a$ pv if $C+\beta a\leqslant C_I$ or $C_I<C=C+\beta a$
C_R	1 if $C\leqslant C_R<C+\beta a$ 0 else	0	1 if $C\leqslant C_R=C_I<C+\beta a$ 0 else
C_I	1 if $C\leqslant C_I=C_R<C+\beta a$ 0 else	0	1 if $C\leqslant C_I<C+\beta a$ 0 else

If the inequality $C\leqslant C_j<C+\beta a$ holds, player j thinks that with probability 1 catastrophe C_j will occur since with the scheduled emission a the critical threshold is passed. The probability for $C_j<C<C+\beta a$ is only defined so that the scope of the definition covers all possible states and choices. Nevertheless, the probability is defined such as to express the idea of player j that although C_j has turned out as a view too pessimistic, $C_j<C$ and any further increase $C<C+\beta a$ will result in a catastrophe. From the results below it is obvious that the specific definition of $C_R<C$ has no consequence.

State k cannot be changed: $P_j(k|k,o,o,o) = 1$ $(j=R,P,I)$. Since no utility functions are known for the three players, we start with linear ones which are simplest to assess. Let the transition from state s and measures (l,a,p) to state t have the utility $U_j(s;\ l,a,p,t)$ for players j=R,P,I.

$$U_R(C,L,l,a,p;\ C+\beta a,M) = c_1 l+c_2 a+c_3 p \quad , \quad (M=L,\tfrac{L}{2}) \quad ;$$

$$U_R(C,L,l,a,p;\ k) = c_1 l+c_2\frac{k-C}{\beta} + c_3 p+c_R \ ;$$

$$U_R(k_4 0,0,0;\ k) = 0;$$

$$U_P(C,L;\ l,a,p;\ C+\beta a,M) = c_4 a \quad , \qquad (M=L,\tfrac{L}{2}) \quad ;$$

$$U_P(C,L,;\ l,a,p;\ k) = c_4\frac{k-C}{\beta} + c_P \quad ;$$

$$U_P(k;\ o,o,o;\ k) = 0 \quad ;$$

$$U_I(C,L;\ l,a,p;\ C+\beta a,M) = c_5 a + c_6 p \quad , \qquad (M=L,\tfrac{L}{2}) \quad ;$$

$$U_I(C,L;\ l,a,p;\ k) = c_5\frac{k-C}{\beta} + c_6 p + c_I \quad ;$$

$$U_I(k;\ o,o,o;\ k) = 0 \quad .$$

The parameters are assumed to have the signs $c_1 \gtrless 0$, $c_2>0$, $c_3<0$, $c_4>0$, $c_5>0$, $c_6<0$. c_j (j=R,P,I) is the additional payoff to player j due to catastrophe and therefore regarded as largely negative. $c_1 \gtrless 0$ reflects the regulator's internal difficulties in setting small standards, $c_2>0$, $c_4>0$, $c_5>0$ the benefits of energy production; $c_3<0$ the damage to the regulator due to pressure exerted on him; and $c_6<0$ the burden of organization. The term $\frac{k-C}{\beta}$ expresses that energy production is only valuable up to the critical amount. Thus the idea is excluded that in the case of a slowly developing catastrophe energy production by combustion of fossil fuel may give additional benefits during the initial stages of the catastrophe.

A play π of the game is given by an infinite sequence

$$\pi = (s^1,l^1,a^1,p^1;\ s^2,l^2,a^2,p^2;\ldots)$$

of states, measures of the regulator, producer, and impactee, respectively. According to the list of transition probabilities, there are only sequences where

$$C^1 \leq C^i \leq C_P \text{ and } L^i \varepsilon \{L^1, \tfrac{L^1}{2}, \tfrac{L^1}{4}, \ldots\} \quad ,$$

and $a^i = \frac{C^{i+1}-C^i}{\beta}$ if $s^{i+1} = (C^{i+1}, L^{i+1})$.

Furthermore if $s^i = k$ then $s^m = k$ for $m > i$. As a first approach we define the utility of a play as the undiscounted infinite sum of the transition utilities:

$$\underline{U}_j(\pi) = \sum_{i=1}^{\infty} U_j(s^i,l^i,a^i,s^{i+1}) \quad .$$

Since the summed-up internal utilities $\Sigma c_1 l^i$ can become infinite we omit them by specifying $c_1 = 0$. Let $(s^1,l^1,a^1,p^1,\ldots)$ denote a play where $s^i = (C^i,L^i)$ and $s^{i+1} = k$.

Then

$$\underline{U}_R(s^1\ldots) = \sum_{j=1}^{i}(c_2a^j+c_3p^j) + c_2\frac{k-C^i}{\beta} + c_3p^{i+1} + c_R$$

$$= c_3\sum_{j=1}^{i+1} p^j + c_2\frac{k-C^1}{\beta} + c_R \quad .$$

In the case of $s^j = (C^j,L^j)$ for $j=1,2,\ldots$

$$\underline{U}_R(s^1,\ldots) = c_3\sum_{j=1}^{\infty} p^j + \lim_{j\to\infty}\frac{C^j-C^1}{\beta} \quad .$$

Admitting $-\infty$ as a payoff then $\underline{U}_R(s^1,\ldots)$ is well defined because of $C^i\leqslant C_P$.

The same argument gives

$$\underline{U}_P(\ ) = c_4\frac{k-C^1}{\beta} + C_P \quad ,$$

$$U_P(\ ) = c_4 \lim \frac{C^j-C^1}{\beta} \quad ;$$

and

$$\underline{U}_I(\ ) = c_5\frac{k-C^1}{\beta} + c_6\sum_{j=1}^{i+1} pj + c_I \quad ,$$

$$\underline{U}_I(\ ) = c_5 \lim \frac{C^i-C^1}{\beta} + c_6\sum_{j=1}^{\infty} p^i \quad ;$$

respectively.

The game is now completely described except for the definition of strategies. For simplification we admit only stationary strategies where the choices depend only on the last state and last measures of the other players.

Definiton: A strategy σ_R of the regulator is a map:

$$\sigma_R : S \to \mathbb{R}$$

such that

$$\sigma_R(C,L) \in M_R(C,L) = \{l\,|\,0\leqslant l\leqslant L\} \quad ,$$

$$\sigma_R(k) = 0 \quad .$$

A strategy σ_P of the producer is a map

$$\sigma_P : \{(s,l) \,|\, s \varepsilon S, l \varepsilon M_R(s)\} \to \mathbb{R} \quad ,$$

such that

$$\sigma_P(C,L,l) \ \varepsilon \ M_P(C,L,l) = \{a \,|\, 0 \leqslant a \leqslant l, \frac{C_P - C}{\beta}\} \quad ,$$

$$\sigma_P(k,0) = 0 \quad .$$

A strategy σ_I of the impactee is a map

$$\sigma_I : \{(s,l,a) \,|\, s \varepsilon S, l \varepsilon M_P(s), a \varepsilon M_P(s,l)\} \to [0,1] \quad ,$$

such that

$$\sigma_I(C,L,l,a) \ \varepsilon \ [0,1] \quad ,$$

$$\sigma_I(k,0,0) = 0 \quad .$$

The sets of strategies are denoted by Σ_j $(j = R,P,I)$.

Due to the list of transition probabilities defined above infinitely many plays can occur. The appropriate σ-algebra over the set Π of all possible plays is defined as the minimal σ-algebra containing all cylinders with finite bases (see M. Loève, 1955, 8.3). Due to the theorem of Tulcea there exist probability measures $P_j(\cdot|\sigma_R,\sigma_P,\sigma_I)$ on this σ-algebra where $P_j(\cdot|\sigma_R,\sigma_P,\sigma_I)$ stems from the iteration of given subjective probabilities.

The payoff function to player j is defined as his high subjective expected utility

$$V_j(\sigma_R,\sigma_P,\sigma_I) = \int \underline{U}_j(\pi)\, dP_j(\pi|\sigma_R,\sigma_P,\sigma_I) \quad (j=R,P,I) \quad .$$

The formalism allows to derive a sharp upper bound for $V_j(\sigma_R,\sigma_P,\sigma_I)$. Due to the definition of the transition probability P_R the set of plays with a component state $s^m = (C^m, L^m)$, such that $C^m > C_R$ has probability $P_R(\cdot|\sigma_R,\sigma_P,\sigma_1) = 0$.

Hence only plays $\pi = (s^1, l^1, a^1, p^1; \ldots)$ have to be considered where a component state s^m either equals (C^m, L^m) such that $C^m < C_R$ or C_R. Hence

$$\underline{U}_R(\pi) = c_3 \sum_{j=1}^{i+1} p^m + c_2 \frac{C_R - C^1}{\beta} + c_R \quad \text{if } C^i \leqslant C_R < C^i + \beta a^i \quad ,$$

or

$$\underline{U}_R(\pi) = c_3 \sum_{j=1}^{i+1} p^j + \lim c_2 \frac{C^j - C^1}{\beta} \quad \text{if } C^j \ll C_R \ (j=1,\dots) \quad .$$

In both cases $\underline{U}_R(\pi) \leqslant c_2 \frac{C_R - C^1}{\beta}$ is obvious. Hence

$$V_R(\sigma_R, \sigma_P, \sigma_I) \leqslant c_2 \frac{C_R - C^1}{\beta} \quad .$$

The analogous argument yields $V_I(\sigma_R, \sigma_P, \sigma_I) \leqslant c_5 \frac{C_I - C^1}{\beta}$ whereas $\underline{U}_P(\pi) \leqslant c_4 \frac{C_P - C^1}{\beta}$ immediately implicates

$$V_P(\sigma_R, \sigma_P, \sigma_I) \leqslant c_4 \frac{C_P - C^1}{\beta} \quad .$$

The bounds are sharp in the sense that strategy triples exist yielding the bounds as payoffs.

Let $\sigma_R(C,L) = L$, $\sigma_P(C,L,l) = \min(l, \frac{C_P - C}{\beta})$, $\sigma_I(C,L,l,a) = 0$.

Then $V_R(\sigma_R, \sigma_P, \sigma_I) = c_4 \frac{C_P - C^1}{\beta}$.

We give examples for V_R and V_P below. If the establishment of the payoffs as expected payoffs over Π were more elaborated (see e.g. J. Kindler, 1971) it would be obvious that we arrive at the same payoffs $V_j : \Sigma_R \times \Sigma_P \times \Sigma_I \to \mathbb{R}$ if we replace the component utility U_I by $U_{I,r}$:

$$U_{I,r}(C,L;\ l,a,p;\ C+\beta a, M) = \begin{cases} c_5 a & \text{if } M = L \ ; \\ c_5 a + \frac{c_6}{v} & \text{if } M = \frac{L}{2} \ ; \end{cases}$$

$$U_{I,r}(C,L;\ 1,a,p;\ k) = U_I(C,L;\ l,a,p;\ k) \ ;$$

$$U_{I,r}(k;\ 0,0,0;\ k) = U_I(k;\ 0,0,0,k) \quad .$$

This remark permits to shorten proofs in the next section.

THE GAME-THEORETIC SOLUTION

Except for two-person zero-sum games or equivalent games, there is no unanimous solution concept. Instead there are a variety. Therefore we shall first give brief definitions of the solution concepts (for a broader discussion see R. Avenhaus and E. Höpfinger, 1978), and later on describe strategy three-tuples satisfying them.

Definition: A three-tuple $(\sigma_R^+, \sigma_P^+, \sigma_I^+) \varepsilon \Sigma_R x \Sigma_P x \Sigma_I$ of strategies is called a (weak) *equilibrium point* if

$$V_R(\sigma_R^+, \sigma_P^+, \sigma_I^+) \geqslant V_R(\sigma_R, \sigma_P^+, \sigma_I^+) \qquad (\sigma_R \varepsilon \Sigma_R) \quad ;$$
$$V_P(\sigma_R^+, \sigma_P^+, \sigma_I^+) \geqslant V_P(\sigma_R^+, \sigma_P, \sigma_I^+) \qquad (\sigma_P \varepsilon \Sigma_P) \quad ;$$
$$V_I(\sigma_R^+, \sigma_P^+, \sigma_I^+) \geqslant V_I(\sigma_R^+, \sigma_P^+, \sigma_I) \qquad (\sigma_I \varepsilon \Sigma_I) \quad .$$

Definition: The payoff vector $(V_j(\sigma_R, \sigma_P, \sigma_I))j = R,P,I$ is called *Pareto-optimal* if there is no other payoff vector $(V_j(\tau_R, \tau_P, \tau_I))$ where $\tau_j \varepsilon \Sigma_j (j = R,P,I)$, such that

$$V_j(\sigma_R, \sigma_P, \sigma_I) \geq V_j(\tau_R, \tau_P, \tau_I) \quad (j=R,P,I) \quad ,$$

and at least one inequality strictly holding.

Definition: Let $(W_R, W_P, W_I) \varepsilon R^3$ denote the point of maximal possible payoffs which is called *bliss point*, i.e. $W_j = \max(V_j(\sigma_R, \sigma_P, \sigma_I) | \sigma_i \varepsilon \Sigma_i (i = R,P,I))$. The payoff vector (v_R, v_P, v_I) is called bliss-optimal if

$$\sum_{j=R,P,I} (v_j - W_j)^2 = \min\left(\sum_j (V_j(\sigma_R, \sigma_P, \sigma_I) - W_j)^2 \,|\, (\sigma_R, \sigma_P, \sigma_I) \varepsilon \Sigma_R x \Sigma_P x \Sigma_I\right)$$

Definition: Let (d_R, d_P, d_I) be a triple of payoffs the players obtain in case they cannot reach an unanimous agreement on the choice of a payoff vector. Then the Nash *solution* is the point (W_R, W_P, W_I) which maximizes the term $(u_R - d_R)(u_P - d_P)(u_I - d_I)$ subject to the requirements $u_j = V_j(\sigma_R, \sigma_P, \sigma_I)$ $(j = R,P,I)$ for some strategy three-tuple and $u_j \geq d_j$ $(j = R,P,I)$.

Definition: A *hierarchic solution* is a triple (τ_R,τ_P,τ_I) consistent of a strategy $\tau_R \varepsilon \Sigma_R$, and two maps

$$\tau_P : \Sigma_R \rightarrow \Sigma_P \quad ,$$

$$\tau_I : \Sigma_R x \Sigma_P \rightarrow \Sigma_I \quad ,$$

such that $V_I\Big(\sigma_R,\sigma_P,\tau_I(\sigma_R,\sigma_P)\Big) = \max\limits_{\sigma_I \varepsilon \Sigma_I} V_I(\sigma_R,\sigma_P,\sigma_I) \quad ;$

$$V_P\Big(\sigma_R,\tau_P(\sigma_R),\tau_I\big(\sigma_R,\tau_P(\sigma_R)\big)\Big) = \max_{\sigma_P \varepsilon \Sigma_P} V_P\Big(\sigma_R,\sigma_P,\tau_I(\sigma_R,\sigma_P)\Big) \quad ;$$

$$V_R\Big(\tau_R,\tau_P(\tau_R),\tau_I\big(\tau_R,\tau_P(\tau_R)\big)\Big) = \max_{\sigma_R \varepsilon \Sigma_R} V_R\Big(\sigma_R,\tau_P(\sigma_R),\tau_I\big(\sigma_R,\tau_P(\sigma_R)\big)\Big)$$

The game has a huge variety of equilibrium points. In the following we give three equilibrium points, the first two of which have Pareto-optimal payoffs, whereas the third is only given as an indicator of the variety of equilibrium points.

Theorem: The tuples of strategies given below are equilibrium points:

1) $\sigma_R^1\,(C,L) := \min\Big(L, \max(0, \frac{C_R-C}{\beta})\Big) \quad ;$

$\sigma_P^1\,(C,L,l) := 1 \quad ;$

$\sigma_I^1\,(C,L,l,a) := 0 \quad .$

The inherent utilities are

$$V_R(\sigma_R^1,\sigma_P^1,\sigma_I^1) = c_2\,\frac{C_R-C^1}{\beta} \quad ;$$

$$V_P(\sigma_R^1,\sigma_P^1,\sigma_I^1) = c_4\,\frac{C_R-C^1}{\beta} \quad ;$$

$$V_I(\sigma_R^1,\sigma_P^1,\sigma_I^1) = \begin{cases} c_5\,\frac{C_R-C^1}{\beta} & \text{if } C_R \leqslant C_I \quad , \\ c_5\,\frac{C_I-C^1}{\beta} + c_I & \text{if } C_R > C_I \quad . \end{cases}$$

2) $\sigma_R^2(C,L) := \min\Big(L, \max(0, \frac{C_I-C}{\beta})\Big) \quad ;$

$\sigma_P^2(C,L,l) := 1 \quad ;$

$$\sigma_I^2(C,L,l,a): = \begin{cases} 0 & \text{if } l = \min(L,\frac{C_I-C}{\beta}) \text{ and } C \leqslant C_I \quad, \\ 1 & \text{if } l \neq \min(L,\frac{C_I-C}{\beta}) \text{ or } C > C_I \quad. \end{cases}$$

The inherent utilities are

$$V_R(\sigma_R^2,\sigma_P^2,\sigma_I^2) = \begin{cases} c_2\,\frac{C_I-C^1}{\beta} & \text{if } C_I \leqslant C_R \quad, \\ c_2\,\frac{C_I-C^1}{\beta} + c_R & \text{if } C_I > C_R \quad. \end{cases}$$

$$V_P(\sigma_R^2,\sigma_P^2,\sigma_I^2) = c_4\,\frac{C_I-C^1}{\beta} \quad ;$$

$$V_I(\sigma_R^2,\sigma_P^2,\sigma_I^2) = c_5\,\frac{C_I-C^1}{\beta} \quad .$$

3) *Keep quiet point*

$$\sigma_R^3(C,L) = 0 \quad ;$$

$$\sigma_P^3(C,L,l) = 0 \quad ;$$

$$\sigma_I^3(C,L,l,a) = \begin{cases} 0 & \text{if } l=0 \text{ and } C=C^1 \quad, \\ 1 & \text{if } l>0 \text{ or } C>C^1 \quad; \end{cases}$$

with utilities $V_j(\sigma_R^3,\sigma_P^3,\sigma_I^3) = 0 \quad (j=R,P,I)$.

Proof: In order to avoid descriptions that are cumbersome but not illustrative we give sketches only.

1) Let $i_R \in \{1,2,\ldots\}$ be defined by $C^1+\beta(i_R-1)L^1 \leqslant C_R < C^1+\beta i_R L^1$.

One can show by iteration on i that

$$C^{i+1}=C^1+\beta iL^1 \,(i=0,1,\ldots,i_R-1),\quad a^i=L^1\,(i=1,\ldots,i_R-1) \quad ,$$

$$a^{i_R}= \frac{C_R-C^{i_R}}{\beta} \quad ,$$

$$C^{i+1}=C_R\,(i=i_R,i_R+1,\ldots), \qquad a^i=0\ (i=i_R+1,i_R+2,\ldots) \quad ,$$

due to the regulator's strategy. Hence

$$V_R(\sigma_R^1,\sigma_R^2,\sigma_R^3)=c_2\sum_{i=1}^{i_R-1} L^1+c_2\frac{C_R-C^{i_R}}{\beta} = c_2\frac{C_R-C^1}{\beta} \quad ;$$

analogously

$$V_P(\sigma_R,\sigma_P,\sigma_I)=c_4\frac{C_R-C^1}{\beta} \quad .$$

In the case of $C_R \leqslant C_I$, (C_R,L^1) will be the state of the play for $i = i_R + 1, i_R + 2, \ldots$ also due to the subjective probability of the impactee. However, if $C_R > C_I$, catastrophe C_I will be the final state resulting in a payoff $c_5\frac{C_I-C^1}{\beta} + c_I$.

The regulator's condition for an equilibrium is obviously satisfied since the strategy triple gives him the maximal possible utility. Just as is obvious, there is no better payoff for the producer with another strategy, and this is also true for the impactee in the case of $C_R \leqq C_I$.

Only $C_R > C_I$ requires more sophistication. Let σ_P' denote a different strategy of the impactee. Then a play π with $\lim C^i \leqq$ $\leqq C_I$ is only possible if the reduction of L^i to its half takes place an infinite number of times. But then $\underline{U}_{I,r}(\pi) = \infty <$ $c_5\frac{C_I-C^1}{\beta} + c_I$. If the reduction of L^1 takes place only a finite number of times then $\underline{U}_{I,r}(\pi) \leqq c_5\frac{C_I-C^1}{\beta} + c_I$. Hence any other strategy cannot yield a better payoff.

2) In the case of $C_I < C_R$ the regulator can only get a better payoff if plays π with states (C^i,L^i) where $C^i > C_I$ occur with a subjective probability greater than zero. But then $\sigma_I^2(C^i,L^i,1,a)=$ $= 1$ infinitely often yielding the payoff $-\infty$ to the regulator. Thus he cannot get a better payoff with a different strategy. Obviously the producer cannot get a better payoff, whereas the impactee gets his maximal payoff.

In the case of $C_I = C_R$ regulator and impactee receive their maximal payoffs, whereas the producer has no better response. In the case of $C_I > C_R$ the regulator may want to escape catastrophe by applying a strategy like the one of the first equilibrium point. But then he is punished an infinite number of times by pressure from the impactee and gets a smaller payoff. Again it is obvious that producer and impactee cannot do better.

3) The impactee's ability to exert pressure infinitely often again makes the strategy triple $(\sigma_R^3,\sigma_P^3,\sigma_I^3)$ an equilibrium point.

The question arises: Which of these equilibrium points yield Pareto-optimal payoffs? The answer can immediately be deduced from the following:

Theorem: The set of payoffs

$$\left\{\Big(V_R(\sigma_R,\sigma_P,\sigma_I),V_P(\sigma_R,\sigma_P,\sigma_I),V_I(\sigma_R,\sigma_P,\sigma_I)\Big)\,\Big|\,\sigma_j\varepsilon\Sigma_j\,(j=R,P,I)\right\}$$

is a subset of the following domain $D\supseteq\mathbb{R}^3$.

1) Let $C_R < C_I < C_P$. Then D consists of all $(x,y,z)\varepsilon\mathbb{R}^3$ such that a pair (p_R,p_I) of real numbers exists such that $o \leqq p_R$, $o \leqq p_I$, $0 \leq p_R + p_I$ and the following inequalities hold:

$$x\leq c_2\frac{C_R-C^1}{\beta}+(1-p_R)c_R \quad ;$$

$$y\leq\bar{c}_4\left\{p_R\frac{C_R-C^1}{\beta} + p_I\frac{C_I-C^1}{\beta} + (1-p_R-p_I)\frac{C_P-C^1}{\beta}\right\} \quad ;$$

$$z\leq c_5p_R\frac{C_R-C^1}{\beta} + c_5p_I\frac{C_I-C^1}{\beta} + (1-p_R-p_I)\left(c_5\frac{C_I-C^1}{\beta} + c_I\right) \quad .$$

2) Let $C_R = C_I < C_P$. Then D consists of all $(x,y,z)\varepsilon\mathbb{R}^3$ which are part of a solution $(x,y,z,p)\varepsilon\mathbb{R}^4$ of the following system of inequalities:

$$0\leq p \leq 1 \quad ;$$

$$x\leq c_2\frac{C_R-C^1}{\beta} + (1-p)c_R \quad ;$$

$$y\leq c_4\left\{p\frac{C_R-C^1}{\beta} + (1-p)\frac{C_P-C^1}{\beta}\right\} \quad ;$$

$$z\leq c_5p\frac{C_R-C^1}{\beta} + (1-p)\left(c_5\frac{C_R-C^1}{\beta} + c_I\right) \quad .$$

3) Let $C_I < C_R < C_P$. Then D consists of all $(x,y,z)\varepsilon\mathbb{R}^3$ which are part of a solution $(x,y,z,p_Ip_R)\varepsilon\mathbb{R}^5$ of the following system of inequalities:

$$0 \leq p_I,\ 0 \leq p_R,\ 0 \leq 1-p_I-p_R \quad ;$$

$$x \leq c_2 p_I \frac{C_I - C^1}{\beta} + c_2 p_R \frac{C_R - C^1}{\beta} + (1-p_I-p_R)\left(c_2 \frac{C_R - C^1}{\beta} + c_R\right) \quad ;$$

$$y \leq c_4 \left\{ p_I \frac{C_I - C^1}{\beta} + p_R \frac{C_R - C^1}{\beta} + (1-p_I-p_R) \frac{C_P - C^1}{\beta} \right\} \quad ;$$

$$z \leq c_5 \frac{C_I - C^1}{\beta} + (1-p_I) c_I \quad .$$

Sketched proof: Let $(\sigma_R, \sigma_P, \sigma_I)$ denote a strategy triple. In the case of $C_R < C_I \leq C_P$ let p_R denote the probability $(P_P(T_R|\sigma_R,\sigma_P,\sigma_I)$ that a play with states (C^i, L^i), $C^i \leq C_R$ will be realized, i.e. T_R is the set of all plays $(s^1, l^1, a^1, p^1, \ldots)$ such that $C^i \leq C_R$ for all component states (C^i, L^i) $(i = 1,2,\ldots)$. Let $p_I = P_P(T_I|\sigma_R,\sigma_P,\sigma_I)$ denote the probability for the set of plays (s^i, l^i, a^i, p^i) $(i = 1,2,\ldots)$ such that $C^i \leq C_I$ for all i, where $s^i = (C^i, L^i)$ $(i = 1,2,\ldots)$ but $C^j > C_R$ for at least one j. Obviously

$$V_P(\sigma_R,\sigma_P,\sigma_I) \leq c_4 \left\{ p_R \frac{C_R - C^1}{\beta} + p_I \frac{C_I - C^1}{\beta} + (1-p_R-p_I) \frac{C_P - C^1}{\beta} \right\} \quad .$$

By definition of the regulator's transition probability, $P_R(T_R|\sigma_R,\sigma_P,\sigma_I) = p_R$, but with probability $1 - p_R$ the catastrophe will occur. Hence

$$V_R(\sigma_R,\sigma_P,\sigma_I) \leq c_2 \frac{C_R - C^1}{\beta} + (1-p_R) c_R \quad .$$

The impactee's probabilities for plays with only state components below C_R, and between C_R and C_I are p_R and p_I respectively. Therefore

$$V_I(\sigma_R,\sigma_P,\sigma_I) \leq c_5 p_R \frac{C_R - C^1}{\beta} + c_5\, {}_I \frac{C_I - C^1}{\beta} + (1-p_R-p_I)\left(c_5 \frac{C_I - C^1}{\beta} + c_I\right) \quad .$$

The proofs for the two remaining cases follow the same line of argumentation. One has only to consider that p is the producer's subjective probability that a play will occur where $C^i \leq C_R = C_I$ for all component states C^i. In the last case p_I denotes the producer's probability for a play with component states not greater that C_I, p_R, the probability for a play with a component state greater than C_I, and all component states not greater than C_R.

Corollary: The first and the second equilibrium point of the last but one theorem have Pareto-optimal payoff vectors. In the case of $C_R > C^1$ and $C_I \geq C^1$ the keep-quiet point has no Pareto-optimal payoff vector.

Proof: Having chosen either $p_I = 1$ or $p_R = 1$ and $p = 1$, immediately verifies that the payoff vectors of the first and second equilibrium points belong to the boundary plane given on the right-hand side of the inequalities of the last but one theorem. Hence the payoff vectors are Pareto-optimal.

Under the given conditions the keep-quiet point is dominated by the first or the second equilibrium point. The results are illustrated by Figures 1 and 2 showing the projection of subset D of the last theorem.

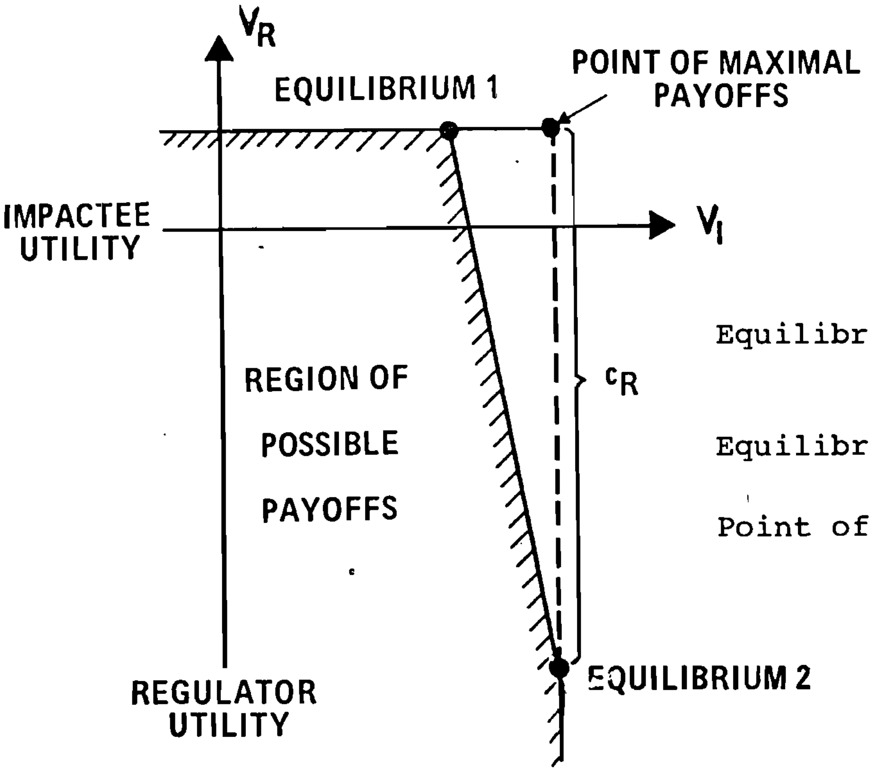
As can be seen from the figures even the combined solution concepts of equilibrium point and Pareto-optimality do not yield an unanimous solution. But what about the remaining solution concepts? In order to discuss them we give the boundary plane of the last theorem after elimination of the parameters for the case of $C_R < C_I < C_P$ by the following equation:

$$\frac{y}{c_4} + \frac{z}{c_I}\,\frac{C_P - C_I}{\beta} + \frac{x}{c_R}\left\{\frac{C_R - C_I}{\beta} + \frac{c_5}{c_I}\,\frac{C_R - C_I}{\beta}\,\frac{C_P - C_I}{\beta}\right\} = \text{constant} \;.$$

Since by assumption c_I and c_R are huge negative numbers the equation is dominated by the first term $\frac{y}{c_4}$. Hence the payoff vector $(C_2\frac{C_R - C^1}{\beta},\ c_4\frac{C_R - C^1}{\beta}, c_5\frac{C_R - C^1}{\beta})$ is either bliss-optimal or very close to the bliss-optimal payoff vector. Hence we can regard it as approximately bliss-optimal.

The same holds for $C_R = C_I < C_P$, and in the case of $C_I < C_R < C_P$ for $(c_2\frac{C_I - C^1}{\beta},\ c_4\frac{C_I - C^1}{\beta},\ c_5\frac{C_I - C^1}{\beta})$.

Without proof we state that the two approximate bliss-optimal points are Nash solutions for $d_j = o$ $(j = R,P,I)$ as soon as the absolute values of c_I and c_R are large enough. This means that the bliss-point concept as well as the Nash solution favor a behavior based on the most pessimistic estimate min (C_R, C_I) of the critical value.



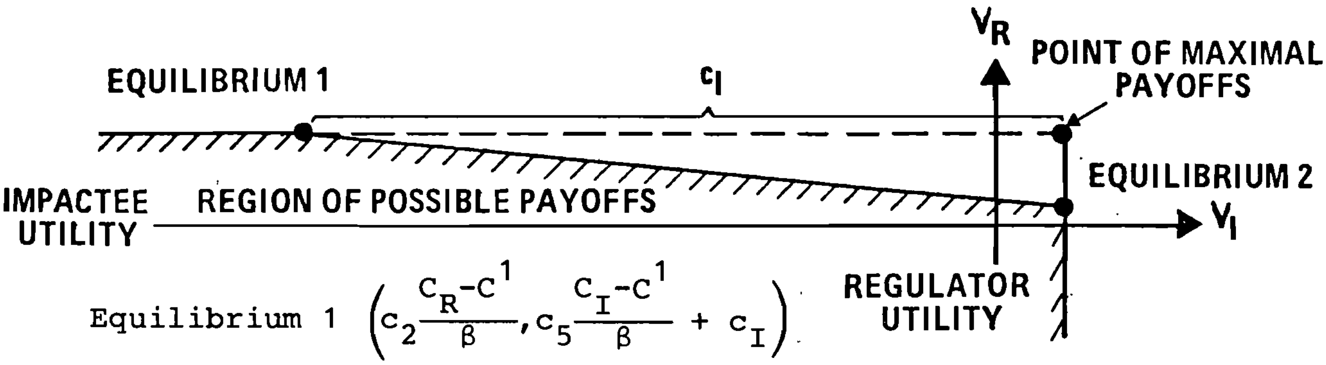
Equilibrium 1 $\left(c_2\frac{C_R-C^1}{\beta}, c_5\frac{C_R-C^1}{\beta}\right)$

Equilibrium 2 $\left(c_2\frac{C_R-C^1}{\beta} + c_R, c_5\frac{C_I-C^1}{\beta}\right)$

Point of Maximal Payoffs

$$\left(c_2\frac{C_R-C^1}{\beta}, c_5\frac{C_I-C^1}{\beta}\right)$$

Figure 1. Payoff diagram for regulator and impactee ($C_R < C_I$).



Equilibrium 1 $\left(c_2\frac{C_R-C^1}{\beta}, c_5\frac{C_I-C^1}{\beta} + c_I\right)$

Equilibrium 2 $\left(c_2\frac{C_I-C^1}{\beta}, c_5\frac{C_I-C^1}{\beta}\right)$

Point of Maximal Payoffs

$$\left(c_2\frac{C_R-C^1}{\beta}, c_5\frac{C_I-C^1}{\beta}\right)$$

Figure 2. Payoff diagram for regulator and impactee ($C_R > C_I$).

The hierarchic solution concept is much more complicated than the preceding ones since it involves maps from strategy spaces into strategy spaces. We circumvent the mathematical optimization problem specifying only the resulting strategies.

Theorem: Let $(\sigma_R^1, \sigma_P^1, \sigma_I^1)$ be the first equilibrium point of the last but one theorem, i.e.,

$$\sigma_R^1(C,L) = \min\left(L, \max(0, \frac{C_R - C}{\beta})\right) \quad ;$$

$$\sigma_P^1(C,L,l) = 1 \quad ;$$

$$\sigma_I^1(C,L,l,a) = 0 \quad .$$

Let $(\tau_R^1, \tau_P^1, \tau_I^1)$ denote a hierarchic solution. Then (τ_R, τ_A, τ_I) defined by

$$\tau_R = \sigma_R^1 \quad ;$$

$$\tau_P(\sigma_R) = \tau_P^1(\sigma_R)\,(\sigma_R \varepsilon \Sigma_R - \{\sigma_R^1\}),\ \tau_P(\sigma_R^1) = \sigma_P^1 \quad ;$$

$$\tau_I(\sigma_R, \sigma_P) = \tau^1(\sigma_R, \sigma_P)\,(\sigma_j \varepsilon \Sigma_j - \{\sigma_j^1\}\,(j=R,P) \quad ;$$

$$\tau_I(\sigma_R^1, \sigma_P^1) = \sigma_I^1 \quad ;$$

is also a hierarchic solution.

Proof: $V_I(\sigma_R^1, \sigma_P^1, \sigma_I^1) = \max_{\sigma_I} V_I(\sigma_R^1, \sigma_P^1, \sigma_I^1)$ since $(\sigma_R^1, \sigma_P^1, \sigma_I^1)$ is an equilibrium point. The next step is the verification of $V_P(\sigma_R^1, \sigma_P^1, \sigma_I^1) = \max_{\sigma_P} V_P\left(\sigma_R^1, \sigma_P, \tau_I(\sigma_R^1, \sigma_P)\right)$. The regulator's strategy σ_R^1 prevents a larger amount than C_R of carbon dioxide in the atmosphere, whereas the producer's utility is the larger the more dioxide is in the atmosphere. Therefore $V_P(\sigma_R^1, \sigma_P^1, \sigma_I^1) = c_4 \frac{C_R - C^1}{\beta} = \max_{\sigma_P, \sigma_I} V_P(\sigma_R^1, \sigma_P, \sigma_I)$, which is even stronger. The last condition is trivially satisfied since $V_R(\sigma_R^1, \sigma_P^1, \sigma_I^1)$ gives the maximal possible utility $c_2 \frac{C_R - C^1}{\beta}$ to the regulator.

It should be remarked that the theorem is independent of whether $C_R < C_I$ or not. It simply states that the regulator is strong enough to push through his standpoint.

The following example serves to illustrate the order of magnitude. Let $C^1 = 6 \cdot 10^{16}g$, $C_I = 18 \cdot 10^{16}g$, $L^1 = 0.2 \cdot 10^{16}g$, $\beta = 0.3$, $c_2 = 0.002\$/g$, $c_4 = 10^{-4}c_2$, $c_5 = 0.7\ c_2$. C^1 is in the order of magnitude of the present amount of carbon dioxide in the atmosphere, and L^1 in the order of magnitude of the present release of carbon dioxide. $\$3.6 \cdot 10^{12}$ is an estimate of the gross world product of 1970. Then production is possible for 200 years and the payoff vector equals ($\$8 \cdot 10^{14}$, $\$8 \cdot 10^{10}$, $\$5.6 \cdot 10^{14}$).

CONCLUSION

The game has been analyzed for different solution concepts. It turns out that the Nash solution and the bliss-optimal concept yield solutions that are basically different from the hierarchic solution. In the case of $C_I < C_R$ where the impactee's view is more pessimistic than that of the regulator, the Nash solution and the bliss-optimum concept, by their tendency to fair bargains, favor the second equilibrium point based on the estimate C_I. Contrary to this the hierarchic solution yields the first equilibrium point which is based on the estimate C_R as critical value.

The results heavily depend on the fact that the summed up component payoffs are not discounted. Thus the impactee can principally push the regulator's payoff down to minus infinity. Actually he cannot exert pressure infinitely often since then he would also receive the payoff minus infinity. Hence this capability to punish or to exert pressure only yields a vastness of equilibrium points. It seems that the results may change substantially if discounting is included. Then the regulator may be able to resist pressure, and on the other side the impactee may be able to afford pressure. Another way would be to assume the game to be stopped as soon as the upper bound L is below a given limit, e.g., if L is less than ten percent of the carbon dioxide produced by the biosphere during one year. Again the question arises whether the impactee can enforce a total release that is less than $\frac{C_I - C^1}{\beta}$.

So far the impactee has been represented as a rational player with a utility function. Another possibility would be to represent him by a response function based on his perception of the regulator's and the producer's decisions, i.e., to prescribe one strategy of the impactee. Then we would actually have a regulator-producer game, and as soltuion concept we might take the hierarchic solution. But which response should we use? Our analysis of the three-person game offers us two responses:

$$\sigma_I^1(C,L,l,a) = \quad 0 \quad ;$$

$$\sigma_I^2(C,L,l,a) = \begin{cases} 0 & \text{if } l = \min(L,\frac{C_I - C}{\beta}) \text{ and } C \leq C_I \quad , \\ 1 & \text{if } l \neq \min(L,\frac{C_I - C}{\beta}) \text{ or } C > C_I \quad . \end{cases}$$

If we assume the first, then the impactee is actually a dummy player. Then equilibrium point one is part of the hierarchic solution. In the case of σ_I^2 however, the hierarchic solution yields the second equilibrium point as can be verified very easily. Thus, the three-person game can provide for ideas how to formalize a response function.

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A DYNAMIC MODEL FOR SETTING RAILWAY NOISE STANDARDS*

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Abstract

This paper describes the application of a multistage game theoretical model to setting noise standards which is illustrated by the case of trains. The problem was structured to match the decision problem which the Environment Agency faced when setting standards for Shinkansen trains. The model considers three players: the regulator (environment agency), the producer (railway corporation), and the impactees (residents along the railway line who suffer from noise). The game has seven stages characterized by the actions of the impactees ranging from petitions to legal litigation. The final stages are the outcomes of a possible lawsuit. The case is either won by the producer or the impactees, or a compromise is reached. Transition probabilities between stages are considered parameters of the game. They depend mainly on the noise level the impactees consider acceptable, the standard set by the regulator, and the actual level of noise emitted. Only the regulator and the producer are active players in the sense that they have a set of choices characterized as standard levels (regulator) and noise protection measures (producer). The impactees are modeled as a response function. Several solutions according to a hierarchical solution concept of the game are derived. In particular, conditions are given under which the regulator or the producer would prefer a compromise solution to awaiting the outcome of the court case. These conditions can be expressed directly as functions of noise levels and transition probabilities, given some simple assumptions about the shape of the utility functions of the regulator and the producer.

A DYNAMIC MODEL FOR SETTING RAILWAY NOISE STANDARDS

1. INTRODUCTION

Since the superrapid "bullet train", the Shinkansen, began operations in Japan in 1964, complaints about train noise have never ceased. Peak noise levels can reach over 100 dB leading to substantial disturbances of residential living. Since the responses of the government and the railway corporation to these complaints have been slow, citizens began to go through various forms of protest, including petitions, organizations, and legal litigation. In 1972 the government asked the railway corporation to take urgent steps against Shinkansen noise. But it was not until 1975 that noise standards (70-75 dB) were issued to force the railway corporation to respond to the citizens' need for quietness. Residents, however, were not contend with these standards and the railway corporation's subsequent attempts at improving sound protection measures. A legal battle between residents and the railway corporation is still going on in which residents ask to reduce Shinkansen noise to a "nondisturbing" level.

In a recent paper (see [1]) the decision process of the Environment Agency and the railway corporation was described and analyzed. In this analysis the need was recognized for more formal methodologies to study decision making involving the conflict between environmental and developmental interests. The present paper is an attempt at developing such a methodology based on dynamic game theoretic models. The purpose of such models is to explore alternative strategies of the conflicting actors in environmental standard setting decisions, and to derive "optimal" strategies depending on the parameters of the game and alternative solution concepts.

Essentially three groups are involved in typical environment-development conflicts: the regulator, the producer (developer), and the impactee (sufferer of pollution). In the case of train noise these groups are an environmental agency (regulator), a railway corporation (producer), and the residents along the line (impactees). Neglecting institutional arrangements, the regulator and the producer are considered single rational players for the purposes of the model. The decisions of the residents are considered (possibly probabilistic) reactions to the decision of the regulator and the producer. Thus the impactee is not modeled as a rational player but rather as a response function. The conflict situation between regulator, producer, and residents is formalized as a multistage two-person game, where a stage is characterized by the action of the residents or a sentence by a court.

2. THE MODEL

Two-person dynamic or multistage games in extensive form (see [2] or [3]) are regarded that are similar to stochastic games. At each stage a component game of perfect information is played that is completely specified by a state. The players' choices do not control only the payoffs but also the transition probabilities governing the component game to be played at the next stage. It is assumed that the regulator and the producer have the same estimates of the transition probabilities.

The states of the game are a subset of

$$\{(i,L)\,|\,i = 1,\ldots,7; \quad \underline{n} \leq L \leq \overline{n}\} \quad ,$$

where i indicates the last action or measure of the residents or the court. L denotes an upper bound for the admitted noise level, $\overline{n}$ the maximum value of noise produced by the train without special sound protection measures, $\underline{n} > o$ the minimum value of noise under which the train can be run under economic considerations, and (1,L) is the first state after construction of the railway line. Hence $(1,L) = (1,\overline{n})$. State (2,L) indicates that a petition has taken place. (3,L) states that the population affected by noise has built up an organization for negotiations with government in order to arrive at a low noise standard. If the negotiations fail the residents can start a lawsuit. This is indicated by (4,L). (4,L) can be followed by states of type (5,L), (6,L), or (7,L). (5,L) stands for a permanent compromise between all parties with upper bound L for noise. (6,L) indicates that the lawsuit was decided in a neutral or positive way for the railway corporation and the government, and (7,L) that the lawsuit was decided in favor of the residents. (5,L), (6,L), and (7,L) are final or absorbing states. See also Figure 1. For each class of states the component game and the transition probability are specified separately.

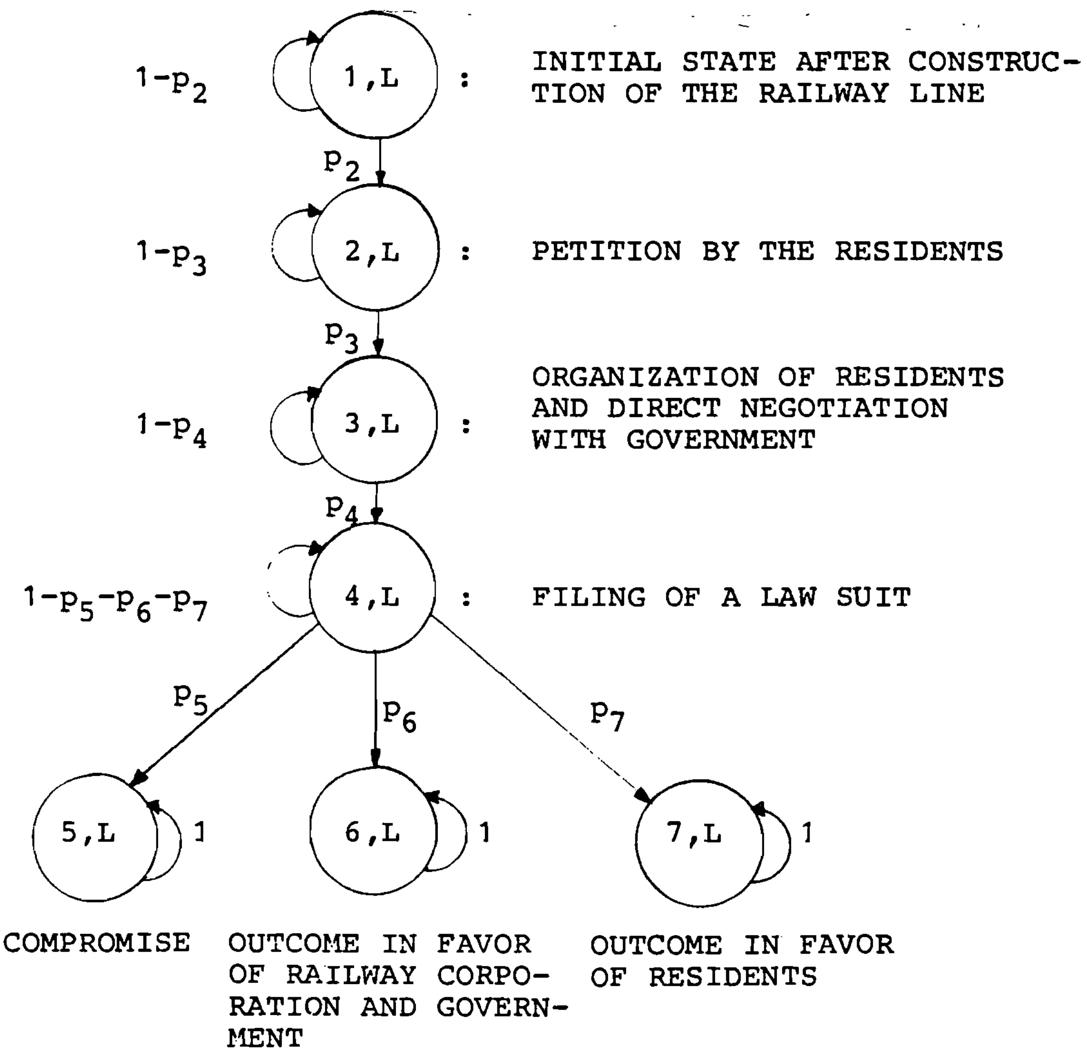

Figure 1. States of the game and transition probabilities (p_i).

The model assumes that the costs and benefits of restricting or increasing noise levels from the train can be expressed as utility functions on noise levels. The utility function of the railway corporation is given as

$$u_P : [\underline{n}, \bar{n}] \to \mathbb{R} \quad ,$$

as long as there is no effective action by the residents. In general, this function will be strictly increasing. In fact, there exists evidence that within reasonable values of $\underline{n}$ and $\bar{n}$ (e.g. 60 and 100 dB, respectively) this function may be linear (see [1]). Thus in some cases it may be possible to express u_P as

$$u_P(n) = n + e \quad ,$$

neglecting a scaling factor.

The utility function of the regulator is also assumed to be defined directly on noise levels:

$$u_R : [\underline{n}, \bar{n}] \to \mathbb{R} \quad .$$

u_R is to reflect a compromise between the economic importance of the train and the noise pollution effects on residents along the line. In the model u_R is assumed to be unimodal with a peak at $\underline{n} \le L^+ \le \bar{n}$. The following argument supports the assumption that u_R is unimodal. Assuming that u_R balances environmental and developmental interests, a crude approximation of u_R could be given by

$$u_R = Wu_P + u_I \quad ,$$

where $W > o$ is an importance weight factor which indicates the relative weight of economic considerations, and u_I is the impactee's utility function. From survey data [4,5] one can infer that the strength of complaints to noise (an indicator of u_I) is approximately quadratically related to noise level. Thus neglecting scaling factors

$$u_I = -(n - \underline{n})^2 + f \quad .$$

Substituting u_I and u_P in u_R gives

$$u_R = W(n + e) - (n - \underline{n})^2 + f \quad ,$$

which is unimodal with a maximum at $L^+ = \frac{W}{2} + \underline{n}$.

In case of the first state $(1,L) = (1,\bar{n})$ the component game is specified as follows. First the regulator chooses his measure $m_R \varepsilon M_R(1,\bar{n})$, where $M_R(1,\bar{n})$ denotes the set of measures available to him. Knowing m_R the producer chooses $m_P \varepsilon M_P(1,\bar{n},m_R)$ where $M_P(1,\bar{n},m_R)$ is the set of measures available to him. M_R and M_P are specified by

$$M_R(1,\bar{n}) := \{l \mid \underline{n} \leq l \leq \bar{n}\} \quad ,$$

$$M_P(1,\bar{n},l) := \{\underline{n} \leq n \leq l\} \quad ,$$

where l denotes the highest level of noise the regulator allows, and n the value of noise generated by operating the railway. The residents' choices are not specified because they are formalized by a response function resulting in special transition probabilities.

A substantial property of the model is the assumption of a threshold $n_I \varepsilon [\underline{n},\bar{n}]$, so that a noise level below n_I is not considered a relevant disturbance of the residents.

Given state $(1,\bar{n})$ only states $(1,\bar{n})$ and $(2,\bar{n})$ can succeed. Regulator and producer believe the transition probabilities to be

$$P((1,\bar{n}) \mid 1,\bar{n},l,n) = \begin{cases} 1 & \text{if } n \leq n_I \\ 1-p_2 & \text{if } n > n_I \end{cases} \quad ,$$

and

$$P((2,\bar{n})\,|\,1,\bar{n},l,n) = 1 - P((1,\bar{n})\,|\,1,\bar{n},l,n)$$

where $p_2 > o$ represents the expert subjective probability that the residents will choose a petition if $n > n_I$. The utilities are given by

$$u_j(1,\bar{n},l,n) = u_j(n) \quad (j = R,P; \quad \underline{n} \le n \le l) \quad .$$

The state $(2,\bar{n})$ can either remain or be replaced by $(3,\bar{n})$ denoting the formation of an organization. We assume the following transition probabilities:

$$P((2,\bar{n})\,|\,2,\bar{n},l,n) = \begin{cases} 1 & \text{if } n \le n_I \\ 1-p_3 & \text{if } n > n_I \end{cases} \quad ,$$

and

$$P((3,\bar{n})\,|\,2,\bar{n},l,n) = 1 - P((2,\bar{n})\,|\,2,\bar{n},l,n) \quad ,$$

where $p_3 > o$. The idea is that $n \le n_I$ is generated either by the regulator $(l \le n_I)$ or by the producer $(n \le n_I < l)$ giving in to the residents' demands. The payoffs are specified by

$$u_j(2,\bar{n},l,n) := u_j(n) \quad (j = R,P; \quad \underline{n} \le n \le l) \quad .$$

In case of a formation of an organization $(3,\bar{n})$ residents will begin negotiations aimed at forcing the regulator to give in and set an acceptable standard. Let the measure set of the regulator and the producer be given by

$$M_R(3,\bar{n}) := \{l\,|\,\underline{n} \le l \le \bar{n}\} \quad ,$$

$$M_P(3,\bar{n},l) := \{n\,|\,\underline{n} \le n \le l\} \quad .$$

Then

$$P((3,\bar{n})|3,\bar{n},l,n) = \begin{cases} 1 & \text{if } l \leq n_I \\ 1-p_4 & \text{if } l > n_I \end{cases} ,$$

$$P((4,\bar{n})|3,\bar{n},l,n) = 1 - P((3,\bar{n})|3,\bar{n},l,n) \quad ,$$

where $p_4 > o$ and $(4,\bar{n})$ denotes the start of a lawsuit. Let

$$u_j(3,\bar{n},l,n) = u_j(n)\,(j = R,P; \quad \underline{n} \leq n \leq l) \quad .$$

Three outcomes of a lawsuit are considered. There is a compromise (5,L) suspending the lawsuit, or a sentence in favor of regulator and producer (6,L), or a sentence in favor of the residents (7,L). Let

$$M_R(4,\bar{n}) = \{l|\underline{n} \leq l \leq \bar{n}\} \cup \{(l,\Lambda)|\underline{n} \leq l \leq \Lambda < \bar{n}\} \quad ,$$

$$M_P(4,\bar{n},l) = M_P(4,\bar{n},l,\Lambda) = \{n|\underline{n} \leq n \leq l\} \cup \{(n,N)|\underline{n} \leq n \leq$$

$$\leq N < \bar{n}, \; n \leq l\} \quad .$$

(l,Λ) indicates that the regulator fixes a bound l for the noise at the current stage and at the same time makes a permanent commitment for a fixed bound Λ in later stages. Λ could be interpreted as a quality standard to be effective permanently after a fixed period of time has passed. For simplicity we assume that Λ becomes effective immediately. Analogously n in (n,N) denotes the actual noise level at the current stage, while N denotes a commitment made by the producer to regard this limit from now on. Let

$$M_C = \{(l,\Lambda,m_P)|(l,\Lambda)\varepsilon M_R(4,\bar{n}), \quad \Lambda \leq n_I, m_P \varepsilon M_P(4,\bar{n},l)\}$$

$$\cup \{(m_R; \quad n,N)|m_R \varepsilon M_R(4,\bar{n}), \quad (n,N)\varepsilon M_P(\bar{n},4,m_R), N \leq n_I\}$$

be called the set of compromise pairs of choices. M_C contains just the pairs (m_R,m_P) of measures guaranteeing to the residents that from now on no noise level greater than n_I will occur. Then we assume

$$P((5,L)|4,\bar{n},m_R,m_P) = \begin{cases} 1 & \text{if } (m_R,m_P)\in M_C \text{ and } L = \min(\Lambda,N) \\ & \text{where } \Lambda := +\infty \text{ or } N := +\infty \text{ in} \\ & \text{case it is not defined} \\ 0 & \text{else} \end{cases},$$

$$P((6,L)|4,\bar{n},m_R,m_P) = \begin{cases} p_6 & \text{if } L = n_R \text{ and } (m_R,m_P)\notin M_C \\ 0 & \text{else} \end{cases},$$

$$P((7,L)|4,\bar{n},m_R,m_P) = \begin{cases} p_7 & \text{if } L = n_I \text{ and } (m_R,m_P)\notin M_C \\ 0 & \text{else} \end{cases},$$

where $\underline{n} \leq n_I \leq n_R \leq \bar{n}$ holds for the maximal noise level n_R fixed by a court sentence in favor of the producer, and $p_6 + p_7$ need not equal 1. Hence

$$P((4,\bar{n})|4,\bar{n},m_R,m_P) = \begin{cases} 0 & \text{if } (m_R,m_P)\in M_C \\ 1-p_6-p_7 & \text{if } (m_R,m_P)\notin M_C \end{cases},$$

The payoffs are specified by

$$u_j(4,\bar{n},m_R,n) = u_j(4,\bar{n},m_R,n,N) = u_j(n)\,(j = R,P;\ \ \underline{n} \leq n \leq l) \quad .$$

State (5,L) means that either the regulator has agreed to take $L \leq n_I$ as the maximal level of noise, or that the producer has bound himself to noise levels not larger than $L \leq n_I$. Let the sets of measures be given by

$$M_R(5,L) := \{l|\underline{n} \leq l \leq L\} \quad ,$$

$$M_P(5,L,l) := \{n|\underline{n} \leq n \leq l\} \quad .$$

Then

$$P((5,L)|5,L,l,n) = 1 \quad .$$

The payoffs are specified by

$$u_j(5,L,l,n) = u_j(n)\,(j = R,P;\ \ \underline{n} \le n \le l) \quad .$$

State $(6,n_R)$ indicates a sentence unfavorable to the residents. Let

$$M_R(6,n_R) = \{l|\underline{n} \le l \le n_R\} \quad ,$$

$$M_P(6,n_R,l) = \{n|\underline{n} \le n \le l\} \quad .$$

Then

$$P((6,n_R)|6,n_R,l,n) = 1 \text{ and } u_j(6,n_R,l,n) = u_j(n)\,(j = R,P) \quad .$$

State $(7,n_I)$ denotes a sentence unfavorable to regulator and producer. Let

$$M_R(7,n_I) = \{l|\underline{n} \le l \le n_I\} \quad ,$$

$$M_P(7,n_I,l) = \{n|\underline{n} \le n \le l\} \quad .$$

Then

$$P((7,n_I)|7,n_I,l,n) = 1 \quad .$$

In the case of a lost lawsuit the producer's and the regulator's utilities change. This is because such a sentence would have much wider reaching consequences than a voluntary agreement to a standard. First of all, implementation time, rules of operation, etc. prescribed in a sentence would mean substantial restriction of freedom to the railway corporation. Secondly, the sentence would most likely be applied throughout the railway network.

Thus the model assumes that

$$U_P(7,n_I,1,n) = u_P(n) + c_P \quad ,$$

where $c_P < o$ is a fixed penalty as a result of the sentence. Also, the regulator stands to lose both in prestige and in lost flexibility if the court should decide in favor of the impactees. Again this loss is expressed in his utility function.

$$U_R(7,n_I,1,n) = u_R(n) + c_R \quad .$$

In the case of $L^+ > n_I$ it appears not unreasonable to assume that $c_j(j = P,R)$ is a negative multiple m_j of $u_j(L^+) - u_j(n_I)$, i.e.

$$C_j = - m_j[u_j(L^+) - u_j(n_I)] \quad , \quad (j = P,R) \quad .$$

A play π of the game is given by an infinite sequence $(s^1, m_R^1, m_P^1; \; s^2, m_R^2, m_P^2; \; \ldots)$ of states and measures. We define the utility of a play π by the discounted infinite sum of the stage utilities

$$\underline{U}_j(\pi) : = \sum_{i=1}^{\infty} \rho^{i-1} U_j(s^i, m_R^i, m_P^i) \quad (j = R,P) \quad ,$$

where $0 < \rho < 1$ is a discount factor.

The game is now completely described except for the definition of strategies and the solution concept. For simplifications we admit only stationary strategies where the choices depend only on the last state and the last measures of the other players.

Definition: A strategy σ_R of the regulator is a map

$$\sigma_R : S \to [\underline{n}, \bar{n}] \quad ,$$

such that

$$\sigma_R(s)\varepsilon M_R(s) \quad (s\varepsilon S) \quad ,$$

where S denotes the set of states.

A strategy σ_P of the producer is a map

$$\sigma_P : \{(s,l)\,|\,s\varepsilon S, l\varepsilon M_R(s)\} \rightarrow [\underline{n},\overline{n}] \quad ,$$

such that

$$\sigma_P(s,l)\varepsilon M_P(s,l) \quad .$$

The sets of strategies are denoted by Σ_R and Σ_P.

For each strategy pair (σ_R,σ_P) a play $\pi = (s^1,l^1,n^1;\ s^2,l^2,n^2;\ \ldots)$ is realized. Since the strategies are stationary, two components (s^i,l^i,n^i) and (s^r,l^r,n^r) are equal as soon as $s^i = s^r$. By the definition of the transition probabilities at most seven states can occur with probability greater than zero and only one will be repeated infinitely often. From this it follows that the set $\Pi(\sigma_R,\sigma_P)$ of possibly realized plays π is finite or denumerable. The probability $P(\pi|\sigma_R,\sigma_P)$ for $\pi\varepsilon\Pi(\sigma_R,\sigma_P)$ is given as an infinite product of the terms $P(s^{i+1}|s^i,l^i,n^i)$ defined above. The payoff of player $j\varepsilon\{R,P\}$ is supposed to be his expected utility of the plays:

$$V_j(\sigma_R,\sigma_P) = \sum_{\pi\varepsilon\Pi(\sigma_R,\sigma_P)} \underline{U}_j(\pi)P(\Pi|\sigma_R,\sigma_P) \quad .$$

The strategies are to be determined according to the following solution concept.

<u>Definition</u>: A hierarchical solution is a pair (τ_R,τ_P) of a strategy $\tau_R\varepsilon\Sigma_R$ and a map $\tau_P : \Sigma_R \rightarrow \Sigma_P$ such that

$$V_P(\sigma_R,\tau_P(\sigma_R)) = \max_{\sigma_P\varepsilon\Sigma_P} V_P(\sigma_R,\sigma_P) \quad ,$$

$$V_R(\tau_R,\tau_P(\tau_R)) = \max_{\sigma_R\varepsilon\Sigma_R} V_R(\sigma_R,\tau_P(\sigma_R)) \quad ,$$

3. THE GAME-THEORETIC SOLUTION

In order to keep the analytical part as small as possible we shall only discuss heuristic equations which, however, can be justified as soon as one establishes the analytical framework in full detail. At least part of it can be found in [7]. Because of the definiton of the component game payoffs and the transition probabilities, the measures $(1,\Lambda)$ and $(1,n_I)$, or (n,N) and (n,n_I), respectively, have the same effect in the case of $\Lambda < n_I$, respectively. This also holds in the case of 1 and $(1,\bar{n})$ or n and $(n,\bar{n})$. Hence, without loss of generality, we can reduce the measure sets $M_R(4,\bar{n},m_R)$ to

$$M_R(4): = \{(1,\Lambda)|n_I \leq \Lambda \leq \bar{n},\underline{n} \leq 1 \leq \Lambda\} \quad ,$$

$$M_P(4,1,\Lambda): = \{(n,N)|n_I \leq N \leq \bar{n},n \leq N,n \leq 1\} \quad .$$

Since then only the states $(1,\bar{n})$, $(2,\bar{n})$, $(3,\bar{n})$, $(4,\bar{n})$, $(5,n_I)$, $(6,n_R)$, $(7,n_I)$ can occur, the states are completely fixed by their first component. We therefore drop the second component in all the terms.

For the rest of the paper let $\Gamma_b (b = 1,\ldots,7)$ denote sub-games of the original game such that b ist the first state. Hence Γ_1 is the original game. $\Gamma_b (b = 5,6,7)$ has only the state b. $\Gamma_b (b = 1,2,3,4)$ covers states b, b + 1,...,7. Though in principle one has to distinguish the strategies for different Γ_b we denote by abuse of notation the reduction of $\sigma_j\varepsilon\Sigma_j$ to Γ_b by σ_j. Let $V_{j,b}$ denote the payoff function for player j in game

Γ_b. Without the simple proof we state that

$$V_{j,7}(\sigma_R,\sigma_P) = \frac{1}{1-\rho}\,(u_j(\sigma_P(7,\sigma_R(7)) + C_j) \quad ,$$

$$V_{j,b}(\sigma_R,\sigma_P) = \frac{1}{1-\rho}\,u_j(\sigma_P(b,\sigma_R(b))) \qquad (b = 5,6) \quad ,$$

for $j = R,P$. Now let $(\sigma_R,\sigma_P)(i) := (\sigma_R(i),\sigma_P(i,\sigma_R(i))(i = 1,\ldots,7)$.
Then

$$\begin{aligned} V_{j,4}(\sigma_R,\sigma_P) = {} & u_j(\sigma_P^1(4,\sigma_R(4))) \\ & + \rho\; P(4|4,(\sigma_R,\sigma_P)(4))V_{j,4}(\sigma_R,\sigma_P) \\ & + \rho\; P(5|4,(\sigma_R,\sigma_P)(4))V_{j,5}(\sigma_R,\sigma_P) \\ & + \rho\; P(6|4,(\sigma_R,\sigma_P)(4))V_{j,6}(\sigma_R,\sigma_P) \\ & + \rho\; P(7|4,(\sigma_R,\sigma_P)(4))V_{j,7}(\sigma_R,\sigma_P) \quad , \end{aligned}$$

where $\sigma_P^1\,(4,1,\Lambda)$ denotes the first component of $\sigma_P(4,1,\Lambda)$. By backward iteration

$$\begin{aligned} V_{j,b}(\sigma_R,\sigma_P) = {} & u_j(\sigma_P(b,\sigma_R(b))) \\ & + \rho\; P(b|b,(\sigma_R,\sigma_P)(b))V_{j,b}(\sigma_R,\sigma_P) \\ & + \rho\; P(b+1|b,(\sigma_R,\sigma_P)(b))V_{j,b+1}(\sigma_R,\sigma_P) \\ & \qquad (b = 1,2,3; \quad j = R,P) \quad , \end{aligned}$$

Three situations are conceivable:

(1) The regulator can enforce his maximum utility;

(2) If regulator and producer have won the lawsuit, the regulator has to offer $l > L^+$ in order to keep the producer from compromising;

(3) If regulator and producer have won the lawsuit, not even the offer $l = n_R$ can keep the producer from compromising.

Though the calculation of the hierarchical solution for three situations is not difficult for any given set of values of the parameters, the derivation of the hierarchical solution as a function of the parameter would require a lot of space. Therefore we consider only the first and the third situations. The two classes of parameters given, however, do not in general exhaust the set of all the parameter values possible.

At first we establish a pair of strategies yielding the maximum utility to the regulator.

Definition: Let $L^+ > n_I$. The vector of real numbers $L^+, n_I, n_R, \rho, p_6, p_7)$ satisfies the compromise condition of player j (C,j) if

$$u_j(n_I) > \frac{1}{1-\rho(1-p_6-p_7)} \{(1-\rho)u_j(L^+) + p_6\rho\, u_j(\min(L^+, n_R)) + p_7\rho[u_j(n_I) + c_j]\}$$

holds.

As can be seen by the formulae above, (C,j) indicates that a compromise is more advantageous to player j.

Theorem: Let $\phi \varepsilon \Sigma_R$ and $\Psi \varepsilon \Sigma_P$ be defined by

$$\phi(i) := L^+ (i = 1,2,3), \quad \phi(5) := \phi(7) := n_I,$$

$$\phi(6) := \min(L^+, n_R)$$

$$\phi(4) := \begin{cases} (n_I, n_I) & \text{if } L^+ \geq n_I \text{ and (C,R) holds} \\ (L^+, L^+) & \text{if } L^+ \leq n_I \text{ or (C,R) is violated} \end{cases}$$

$$\Psi(i,1) := 1 (i = 1,2,3,4,6,7) \quad ,$$

$$\Psi(4,1,\Lambda) := (1,\Lambda) \quad .$$

Then (ϕ,Ψ) yields the maximal utility to the regulator:

$$V_R(\phi,\Psi) = \sup_{\Sigma_R\times\Sigma_P} V_R(\sigma_R,\sigma) \quad .$$

In order to avoid a lengthy and not instructive proof we only give the idea of the proof. First let $L^+ \leq n_I$. Because of the definition of $U_R(i,m_R,m_P)$ the inequality $U_R(i,m_R,m_P) \leq u_R(L^+)$ holds for all possible states i and measures m_R and m_P. Hence $V_R(\sigma_R,\sigma_P) \leq \frac{1}{1-\rho} u_R(L^+)$ for each $(\sigma_R,\sigma_P)\varepsilon\Sigma_R \times \Sigma_P$. But $V_R(\phi,4) = \frac{1}{1-\rho} u_R(L^+)$ because of $\phi(1) = L^+$ and $P(1)1,(\phi,\Psi)(1) = 1$. Now let $L^+ \geq n_I$. Obviously $V_{R,j}(\sigma_R,\sigma_P) \leq V_{R,j}(\phi,\Psi)\,(j = 5,6,7)$. Then (σ_R,σ_P) with $(\sigma_R,\sigma_P)(i) = (\phi,\Psi)(i)\,(i = 5,6,7)$ maximizes $V_{R,4}(\sigma_R,\sigma_P)$ if $(\sigma_R,\sigma_P)(4) = (\phi,\Psi)(4)$ under consideration of the compromise condition (C,R). Hence $V_{R,4}(\sigma_R,\sigma_P) \leq V_{R,4}(\phi,\Psi)$. The final step of the backward iteration yields $V_R(\sigma_R,\sigma_P) \leq V_R(\phi,\Psi)$ for each pair $(\sigma_R,\sigma_P)\varepsilon\Sigma_R \times \Sigma_P$.

If Ψ is an optimal response to ϕ , i.e. $V_P(\phi,4) = \sup_{\Sigma_P} V_P(\phi,\sigma_P)$, it is not important to derive a hierarchical solution since the regulator can enforce his maximum payoff.

<u>Definition</u>: The payoff vector $(V_R(\sigma_R,\sigma_P),\ V_P(\sigma_R,\sigma_P))$ is Pareto-optimal if there is no other strategy pair $(\sigma_R',\sigma_P'\varepsilon\Sigma_R\times\Sigma_P)$ such that $V_j(\sigma_R,\sigma_P) \leq V_j(\sigma_R',\sigma_P')\,(j = R,P)$ and that at least one inequality is strict.

<u>Theorem</u>: Let $(\phi,\Psi)\varepsilon\Sigma_R\times\Sigma_P$ be defined as in the preceding theorem. Then Ψ is an optimal response to ϕ, i.e. $V_P(\phi,\Psi) \geq$ $\geq V_P(\phi,\sigma_P)(\sigma_P\varepsilon\Sigma_P)$, and $V_R(\phi,\Psi) > V_R(\sigma_R,\sigma_P)(\sigma_R\varepsilon\Sigma_R,\ \sigma_P\varepsilon\Sigma_P)$ if one of the following conditions holds

(i) $L^+ \leq n_I$;

(ii) $L^+ \geq n_I$ and (C,R);

(iii) $L^+ \geq n_I$ and not only (C,R) but also (C,P) is violated.

In these cases $(V_R(\phi,\Psi),V_P(\phi,\Psi)$ is a Pareto-optimal payoff vector.

<u>Sketched proof</u>: Case (i): Because of $\phi(1) = L^+ \leq n_I$ $P(1|1,(\phi,\sigma_P)(1)) = 1$. Hence $V_P(\phi,\sigma_P) = \frac{1}{1-\rho} u_P(\sigma_P(1,L^+))$ where $\sigma_P(1,L^+) \leq L^+$ is maximized by Ψ. In order to obtain a greater payoff $V_P(\sigma_R,\sigma_P)$ for one stage at least L^+ has to be replaced by $n > L^+$. But then the regulator's payoff is smaller because of $u_R(n) < u_R(L^+)$.

Case (ii): $P(5|4, (\phi,\sigma_P)(4)) = 1$ because of $\phi(4) = (n_I,n_I)$. By backward iteration evaluating $V_{P,i}(\phi,\sigma_P)$ $(i = 5,4,3,2,1)$ one immediately sees that Ψ maximizes $V_P(\phi,.)$. The proof of the Pareto-optimality relies on the fact that only strategies σ_R with $\sigma_R(i) = \phi(i)$ $(i = 1,\dots,5)$ give maximal payoff to the regulator. The verification of this fact requires a lengthy and uninstructive discussion which we therefore omit.

Case (iii): Given ϕ the assessment $\Psi(i,l) := 1$ $(i = 5,6,7)$ belongs to an optimal response for all values of the parameters. Because of $\phi(4) = (L^+,L^+)$ and $L^+ > n_I$ a strategy σ_P maximizing $V_{P,4}(\phi,.)$ takes either the value $\sigma_P(4,L^+,L^+) = (n_I,n_I)$ or the value (L^+,L^+). Since (C,P) is violated the second assessment yields a larger utility. Hence Ψ maximizes $V_{P,4}(\phi,.)$. Then obviously Ψ maximizes $V_{P,i}(\phi,.)$ $(i = 3,2,1)$. The Pareto-optimality of $(V_R(\phi,\Psi), V_P(\phi,\Psi))$ can again be verified by changing some values of $(\phi,\Psi)(i)$ proving that they reduce the regulator's payoff.

If (C,P) holds and (C,P) is violated the strategy Ψ is generally not an optimal response of ϕ. The situation can arise where the regulator by reduction of his own payoff can force the maximizing producer to a no-compromise strategy. In order to keep the analytical part small we only treat a special case where this situation cannot arise.

<u>Definition</u>: The vector $(\underline{n},n_I,n_R,\bar{n},\rho,p_6,p_7)$ satisfies the strict compromise condition (SC) if

$$u_P(\underline{n}) > \frac{1}{1-\rho(1-p_6-p_7)}\{(1-\rho)u_P(\bar{n}) + p_6\rho u_P(n_R) + p_7\rho[u_P(n_I) + c_P]\}$$

holds.

(SC) can be interpreted by the way that the utmost offer and threat of the regulator cannot match the value of a compromise for the producer.

Theorem: Let (SC) hold. A hierarchical solution (τ_R, τ_P) is given by $\tau_R = \phi$ and $\tau_P(\sigma_R) = \gamma \varepsilon \Sigma_P$ for each $\sigma_R \varepsilon \Sigma_R$ where

$$\gamma(i,1) := 1 | (i = 1,2,3,5,6,7) \quad ,$$

$$\gamma(4,1,\Lambda) := (\min(1,n_I), n_I) \quad .$$

Sketched proof: Because of (SC) the second component of $\sigma_P(4,\sigma_R(4))$ equals n_I for any optimal response σ_P of any $\sigma_R \varepsilon \Sigma_R$. By backward iteration one immediately sees that γ is an optimal response of each $\sigma_R \varepsilon \Sigma_R$, i.e. $V_P(\sigma_R, \sigma_P) \leq V_P(\sigma_R, \gamma)$. $V_{R,5}(\cdot,\gamma)$ is maximized by ϕ and, more generally, $V_{R,i}(\cdot,\gamma)\,(i = 4,3,2,1)$ as one can see by backward iteration.

Remark: In case of $L^+ > n_I$ and (SC) but violated (C,R) the regulator generally does not obtain the possible maximum payoff

$$V_R(\phi,\Psi) \gneqq V_R(\phi,\gamma) \quad .$$

Part of the results can be given in a more illustrative way. In the case of $n_I < L \leq n_R$ let

$$c_j = - m_j(u_j(L^+) - u_j(n_I)) \qquad (j = R,P) \quad .$$

m_j is assumed to be a constant positive factor. It specifies the weight of the severe consequences of a noise reducing sentence because afterwards the sentence has to be regarded for all other noise-producing activities. A short calculation yields that (C,j) is equivalent to

$$m_j p_7 \rho > 1 - \rho(1 - p_6) \qquad (j = R,P) \quad .$$

The second theorem implies that in the case of $n_I < L^+ \leq n_R$ and $m_R p_7 \rho > 1 - \rho(1 - p_6)$ the regulator prefers the compromise: $\phi(4) = (n_I, n_I)$. In case of $n_I < L^+ \leq n_R$ and $m_j p_7 \rho \leq 1 - \rho(1 - p_6)$ $(j = R,P)$, however, the lawsuit will result in a sentence: $(\phi,\Psi)(4) = (L^+, L^+; \; L^+, L^+)$.

An elementray calculation shows that the expected duration d of the lawsuit is $d = \frac{1}{p_6 + p_7}$. Given d, condition (C,j) is equivalent to

$$\frac{1}{d} \geq p_7 > \frac{1-\rho+\rho\frac{1}{d}}{\rho(m_j+1)} \qquad (j = R,P) \quad .$$

The following example illustrates the relevance of the results. Let $d = 4$ years, $\rho = 0.9$, and $m_R = m_P = 10$. Then $(C,j)(j = R,P)$ is approximately given by $p_7 > 0.03$. Hence a lawsuit should only be executed if the probability for a sentence in favor of the residents in one year is not greater than three percent. If $p_7 = 0.03$ then the probability of such a sentence being pronounced at all is $dp_7 = 0.12$.

4. CONCLUSIONS

A main element of the model is the consideration of the impactees' reactions in standard setting. Under certain assumptions the model could identify the important areas in the decision process of the regulator and the producer. In particular the decision about offering and accepting or rejecting a compromise turned out to be of crucial importance. This decision could be determined as a function of the model parameters in which the subjective probabilities of the outcome of the court proceedings can play a major role.

Model limitations include the "short-sightedness" of the impactees' response which only covers present standards and noise levels. Consequently the strategies of the regulator and the producer do not include commitments for later time periods, e.g. in the form of quality standards. The model results indicate, however, that such extensions are feasible, although at a substantially greater effort. For example, strategies could be in the form of long-term noise reduction plans instead of short-term standards, and impactees' responses would take into account the nature of these plans.

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IV.

THE BIOLOGICAL BASIS FOR STANDARD SETTING

THE BIOLOGICAL BASIS FOR STANDARD SETTING FOR ENVIRONMENTAL POLLUTION: A CRITIQUE

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Technical Report No. 11 written for the Volkswagenwerk Foundation Project, "Procedures for the Establishment of Standards"

The Biological Basis for Standard Setting for Environmental Pollutants: A Critique

L.A. Sagan and A.A. Afifi

Abstract

In this paper, an attempt is made to review some of the principles that underlie physiological responses following exposure to environmental agents. The review takes the form of the comparison of the health effects from two often competing fuel cycles, nuclear and coal. In addition, efforts will be reviewed to assess the current literature of the health effects of energy systems. This review will not be comprehensive, but will rather indicate briefly those areas where there is general agreement and those where the data are madequate.

Although each individual pollutant, whether it be chemical, biological, radioactive, gaseous or solid, has its own peculiarities which requires individual consideration, there are underlying principles which are useful in evaluating and understanding all of them. It was intended, that, by focussing on these two energy systems, theses principles, their uses and abuses could be illustrated.

The Biological Basis for Standard Setting for Environmental Pollutants: A Critique

L.A. Sagan and A.A. Afifi

1. Introduction

The process of standard setting for environmental pollutants requires that there be some trade-offs between the often conflicting needs of economics, social welfare, preservation of the environment and health. Of these, possibly the most difficult to assess is the last. To begin with, no universally acceptable definition of health exists, nor do we know exactly how to measure health, or deviations from health. Furthermore, as techniques for measurement of chemical pollutants both in the environment and in biological tissues become more refined, more chemical species are detected in human tissues. Whether the mere *presence* of potentially toxic substances is considered hazardous is sometimes controversial, particularly when exposure to larger quantities is known to be toxic, or when effects are known to occur only after many years. For example, analyses of drinking water in the United States have recently revealed the presence of dozens of chemical species, many of which are known to be toxic to animals or humans at higher concentration. The responsible administrator knows that the costs of detecting the source of these pollutants, regulating and restricting their discharge is enormous to say nothing of the industrial costs which may be attendant upon such restrictions. When he turns to the toxicologist or public health specialist for advice, he is likely to be confronted with contradictory, confusing and often incomplete data. What is he to do?

In this paper, we attempt a review of some of the principles that underlie physiological responses following exposure to environmental agents. The review takes the form of a comparison of the health effects from two often competing fuel cycles, nuclear and coal. Although each individual pollutant, whether it be chemical, biological, radioactive, gaseous or solid, has its own peculiarities which requires individual consideration, there are certain underlying principles which are useful in evaluating and understanding all of them. It was our hope, that, by focusing on these two energy systems, we could highlight and illustrate these principles, their uses and abuses.

Historically, technologies have been allowed to develop solely on the basis of market forces. Since market forces do not include costs for the health effects imposed on the public by such systems (the "externalities"), they could be conveniently ignored. Health effects to industrial employees have never been compensated at their full market value, but would be, in the majority of industrial activities, a small cost of production. Furthermore, even if interest in health costs had been keener, information simply was not available with which to make accurate assessment of these costs. Such evaluation requires a fairly sophisticated society with an enormous information system of vital statistics, medical skills and investigative ability. Unfortunately, necessary information and knowledge with which to make these judgements is still incomplete, but has been markedly increased only during recent decades. Furthermore, the development and availability of the computer enormously enhanced the ability to manage such vast quantities of data. These capabilities have only recently become available.

Secondly, although we have managed to get along without consideration of health or other environmental costs in the past, we cannot any longer ignore the consequences of such a policy. Trial and error alone simply will no longer do. One wonders if, had the health costs of automobile transportation been anticipated, would we have developed such a technology in the way that we have. No longer are we willing to introduce new chemical or pharmacologic agents without some consideration of possible toxic effects and prior testing. The requirements that environmental impact statements be written for Federal projects in the U.S. institutionalized the requirements for some prior thought of potential environmental consequences and alternatives of new systems. Although it has not been proven that such forethought will have benefits, that is at least the reasonable hope.

Lastly, consideration of the health and environmental effects on energy systems is urged upon us by recognition of the enormous increase in world energy production likely in the near future. Assuming a world population of 12 billion and a per capita consumption of 5 kW, energy requirements could increase almost 10-fold in the next 100 years. Clearly, energy options must be developed in such a way as to optimize health and environmental costs as well as the economic costs of such development.

In this paper, we will review efforts to assess the current literature of the health effects of energy systems. This review will not be comprehensive, but will rather indicate briefly those areas where there is general agreement and those where the data are inadequate, with references to the literature for those who wish to pursue the issues. The literature is not extensive. Charpentier [1,2], in reviewing 159 energy models, found only 13 which incorporated some estimate of environmental damages and only two which contained a health effect evaluation.

Comprehensive study of health effects of an energy system involves several elements. The first is an estimate of the emissions or releases from various portions of the fuel cycle as shown in Fig. 1. This is followed by some estimate of the resulting dose to the population, and is in turn followed by an estimate of the health effects which result from such exposures. The first of these steps is methodologically the simplest. Within the second stage, dispersion, enormous problems are encountered. When emissions are to air, as from a stack, the variety of meteorological conditions that may occur between release and exposure, particularly when the exposed population is at some great distance, make modeling very difficult. Furthermore, during transportation interactions among chemical species may occur that are not well understood. Photochemical smog is an example of these. Emissions may also occur into waterways, resulting in contamination of drinking water, or of the food fish taken from that water. When that water is used for irrigation of agricultural land, pollutants may find their way into the human food chains. These food chains may be highly complex and unusual concentrations may occur unexpectedly. The very high concentraion of fallout radiocesium on arctic lichens and the further concentration of cesium within the tissues of reindeer for whom the lichens are a dietary mainstay led to relatively high body burden among Laplanders for whom, in turn, the reindeer is an important nutritional source.

Estimating damage functions is possibly the most difficult of the functions shown in Fig. 1. Because an understanding of the relationship and uncertainties in environmental health depends so strongly on damage functions, the next section deals in some length with the subject of establishing dose response relationships in human populations.

The last factor shown in Fig. 1 is trauma or accidents in which injuries occur due to falls, burns, explosions etc. Note that in both the case of trauma and chemical emissions, persons occupationally involved are likely to have the greatest exposure to risk.

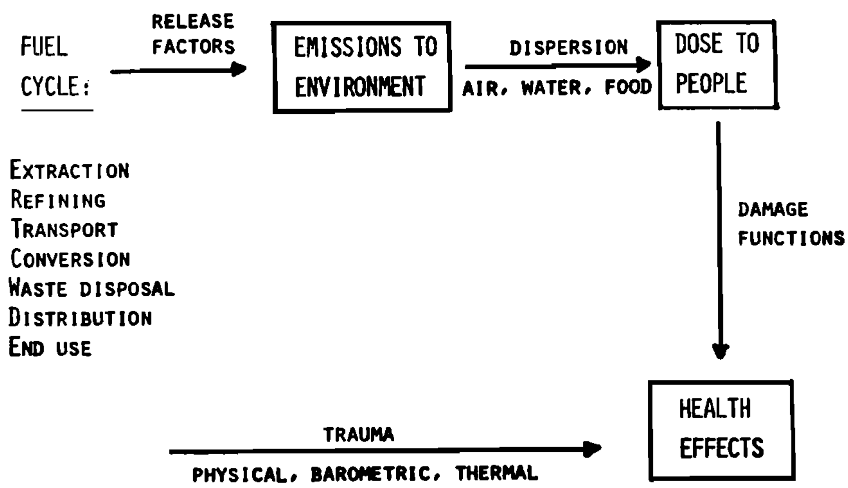

Fig. 1: Elements of Health Effects Estimation

2. Establishing Causal Relationships

Evidence of causality in disease induction derives from a number of sources including animal experimentation and human experiences, both of which are essential. Human observations include accidental exposures, therapeutic exposures, industrial or occupational experiences, and epidemiological studies. The reader should be aware of certain principles which constrain epidemiologic investigation.

2.1 Long latent period

The phrase "latent period" refers to the interval between exposure and appearance of disease. The fact that this interval may be decades enormously complicates the life of the epidemiologist. Examples of such latency are the cancers appearing in uranium miners 20 years following initial exposure, or lung cancers now appearing in asbestos workers whose industrial exposures occurred during World War II. Still another example which might not have been uncovered but for the unusual character of the response is the recent discovery of rare vaginal cancers among young women whose exposures to diethylstilbestrol occurred during intrauterine life some 20 years earlier, a drug with which their mothers had been treated during pregnancy.

Rarely are records or memories adequate to document such remote exposures. Furthermore, because of the great mobility of people, it cannot be assumed that persons residing in a certain community under study had lived there throughout the period of interest. Nor, with rapidly changing industrial practices, can it be assumed that current conditions of exposure had existed during earlier decades. One may then be currently observing effects which are totally unrelated to present environmental conditions but rather to conditions of a distant past for which no records exist. Shocking as it may seem, interest in the environment is very recent and observations of environmental indices frequently do not exist beyond a very few years.

2.2 Non-specificity response

Those diseases which occur in response to environmental exposures, unlike many infectious diseases, are in no way characteristic, i.e. they are clinically indistinguishable from those which occur spontaneously. The leukemia which occurs in atom bomb survivors, for example, is in no known way different from leukemia which occurs in non-radiation exposed persons. To detect such effects, then, requires comparison of the frequency of disease in exposed populations with the frequency of disease in non-exposed populations. The difficulty here is that, for unknown reasons, the frequency of disease varies remarkably from one community to another, even when the populations appear to be quite similar in all known respects and where proper adjustments for age, sex, and race have been made. The problem posed for the epidemiologist then is to find comparable populations which differ only in respect to their exposure. In fact, he cannot often be certain that differences in disease are the result of exposure to a suspect agent or to some unsuspected factor.

It is of considerable benefit in this respect if the disease entity is rare or unusual. For example, the angiosarcoma of the liver now known to occur among persons working with vinyl chloride first came under suspicion when only three cases were found in a small industrial plant. The physician was aware that the probability of a cluster of three cases in such a small population on the basis of chance alone was extremely small. One of the consequences of this situation, of greater interest to lawyers than to epidemiologists, is that one can never be certain in the individual case whether the outcome is the result of a known exposure or might have occurred anyway. All that the epidemiologist can offer is some estimate of the probability that such an exposure was responsible. Legal requirements of proof have been considerably broadened in such cases.

2.3 Knowledge of dose

In order to make some estimate of the effect of exposure to an environmental agent, it is not only necessary to know *whether* such exposure has taken place, but also necessary is some estimate of the amount of exposure. In the matter of environmental pollutants, it is rare that one is comparing an exposed population with a non-exposed population, but rather it is a matter of comparing a more heavily exposed population with a less exposed population, and so the need to estimate dose. A number of obstacles stand in the way of dose-estimation. Some of them are as follows.

Our measures of ambient concentration may not reflect exposure to the individual. A useful example is community air pollution measurements which are typically measured at one or two monitoring stations. Such crude measurements may be totally inaccurate measures of dose for those who are indoors, or for those in distant suburban surroundings or for the traffic policeman standing on a heavily polluted intersection.

Furthermore, we cannot be certain whether it is the peak concentration reached that is important, the average daily dose, average yearly dose or the geometric mean dose. (This leads to the question of dose-rate which will be addressed below in a consideration of radiation effects.) Even this ignores the problem of the faulty measurement of the agent at interest. There is some evidence that the technology for ambient sulfate measurements was badly in error until recently, throwing older measurements into some disrepute. Recalling the above paragraphs on latent periods, it is the measurement of dose 20 years age rather than current measurements that are needed to explain today's morbidity when considering chronic disease induction.

2.4 Synergistic effects

Scientists generally prefer to examine effects of single agents on health parameters. To do so is difficult enough without attempting to unravel the enormously more complicated problems of experimental design when two interacting agents are under study, yet there is fairly good evidence that such synergistic or additive effects do occur (a synergistic effect is one in which the total effect is greater than would be expected on the basis of a summation of two independent effects). In the absence of good evidence of health effects of either SO_2 or particulates, considerable suspicion has fallen upon some interactions between these two elements of air pollution. Better examples of synergy are the established links between cigarette smoking and both uranium mining and asbestosis. The data appears to show a greater than additive effect in the induction of lung cancer among cigarette smokers who are engaged in either of those occupations.

These examples of synergy provide warning signals for unexpected effects that might not be suspected on the basis of studies of individual agent alone, but there is still another phenomenon that further compounds the complexity of the problem and that is the possibility of interference or protective effect of one agent against another. The use of a large number of therapeutic agents are based on this principle. Although no example of interference among agents commonly thought of as pollutants is known, there has been little search for such effects which in all likelihood do exist.

3. Genetic Effects

The foregoing discussion has been fairly extensive reflecting both the authors' interest as well as the breadth of our current state of knowledge. The following discussion of genetic effects will be short for the same reason, but the subject cannot be ignored altogether.

Many agents are known to be mutagenic in animal or other test systems, yet none of these has been shown conclusively to have produced genetic effects in humans. The latter is not taken to mean that humans are peculiarly resistant to mutagenic agents, but rather that the demonstration of mutagenesis is peculiarly difficult. Unlike somatic effects, i.e. those wich occur in the animals who are themselves exposed, the manner in which genetic effects might manifest themselves are unknown. Furthermore, since the majority of mutations are of a recessive character and must occur in both mates to appear in one half of the offspring, the probabilities of demonstrating such effects are vanishingly small. There are occasional reports of cytogenetic changes in circulating lymphocytes of persons exposed to suspect mutagens, but whether or not these are viable and transmissible to offspring is not known. Something more will be said about radiation mutagenesis below.

4. In the Matter of Proof

Under these conditions of uncertainty, what guidelines are available to support a contention of a causal relationship between exposure to an environmental agent and some specific human health effects? There are three: First, that observations be replicated under a large number of conditions, thereby minimizing the likelihood that any one observation be the result of unsuspected variables. For example, the appearance of leukemia in human populations exposed to radiation in both Hiroshima and Nagasaki as well as in therapeutically radiated populations provides indisputable evidence of such an effect. Secondly, a graded dose-response relationship adds important weight to such an observation and such is the case for radiation induced leukemia. Whether or not such a relationship is linear or not is another question that needs not to be discussed here, but is not crucial to the usefulness of some form of gradation. Thirdly, supporting animal data adds strong weight to the claim of causality. Here, some judgement is necessary in extrapolating from the animal data to humans. For example, some mouse strains exposed to radiation develop ovarian cancers. Humans exposed to radiation have not been known to develop ovarian cancers; nevertheless, evidence that radiation is carcinogenic in animals strongly supports the evidence that radiation is carcinogenic in humans, even though not in precisely the same way.

5. Damage Functions

Building on the previous general introduction, we will now turn to consider two specific damage functions: those from radiation exposure, and subsequently, from the emissions from the combustion of coal.

In spite of some uncertainties, evaluation of the health effects of the nuclear power industry is a fairly straightforward operation. Simplicity arises from the fact that there is only a single toxic effluent which requires consideration, i.e. radiation, and secondly because the toxicity of radiation exposure is, although by no means complete, probably better understood than that of any other environmental agent. In turn, reasons for this understanding are multiple, not the least of which is the very large amounts of money that became available for supporting radiation effects research at the end of World War II. Also, radiation is, for the experimentalist, a very easy tool with which to work. Dose can be easily and precisely administered and measured, facilitating animal work. Human studies involving survivors of the atomic bombings and radiation accidents, occupational exposures, and also the thousands of persons treated with radiotherapy permitted thorough analysis of human radiation toxicity. Furthermore, there has been an enormous effort by several organizations, both national and international, to carefully analyse and assess the significance of this data. Their efforts, in which they reach a surprising degree of consensus, simplifies the task greatly. That is not to say that there are no problems. The most frustrating of these problems is the question of the dose-response relationship. For those not familiar with the concept, the latter can be rephrased in the following questions: How much radiation does it take to produce an effect? Does raising or lowering the dose at all levels have a proportional effect, i.e. a linear relationship, or are there deviations from this? What happens if the dose is protracted over a longer time or is fractionated?

For radiation, much of this information is available *at high doses*. By "high" is meant from a few hundred to a few thousand times "background" or natural radiation levels. At those doses, the data is consistent with a number of dose-response models, including a proportional model in which the line passes through the origin. The implication of this interpretation would be that *any* radiation exposure, no matter how slight, would have some small but significant effect. In fact, that is the interpretation that is most widely accepted, not because it better explains or "fits" the data, but because it is considered by most (but not all) to be the most prudent assumption, prudent in the sense that it does not ignore the possibility that low doses *may* produce some effect.

A second assumption underlying the usual assessment of risk from radiation exposure is that dose *rate* has no effect. This is almost certainly an incorrect assumption. The judgement that it is in error arises from widespread animal studies which strongly support the likelihood that dose rate does have an effect [3-5]. Furthermore, evidence that human response to environmental, chemical or nutritional agents is generally influenced by dose rate reinforces the suspicion of this assumption. Consider the nature of a sudden exposure to intense sunlight or alcohol or aspirin. In any of these examples, intemperate exposure can be harmful or lethal whereas the same dose spaced over a longer period of time can be harmless or beneficial.

Why then, is such an assumption made? Again, it is a matter of prudence: Since there are no observations on cancer frequency among human populations exposed to high doses at low dose rates (no such populations exist), the high dose rate data, i.e. radiotherapy, atomic bomb survivors is used even though the populations to whom these risk estimates will be applied will be exposed at very low dose rates, thus leading to what is likely to be a gross exaggeration of risk.

Keeping in mind the above assumptions, just what are the risk estimates? Several organizations have assessed the available data and arrived at such estimates: They are, the United Nations Scientific Committee on the Effects of Ionizing Radiation, UNSCEAR, the U.S. National Academy of Sciences, and the International Committee on Radiation Protection, ICRP. The estimates of the National Academy of Sciences (BEIR Report) are as follows [5]:

(i) It is estimated that exposure of the parents to 170 mrems per year (or 5 rems over the 30 years of the usual reproduction period) would cause in the first generation between about 150 and 3,600 serious genetic disabilities per year in the U.S. population, based on 3.6 million births per year.

(ii) It is estimated that the same exposure of the U.S. population as above *could* cause from roughly 3,000 to roughly 15,000 deaths from cancer annually, with 6,000 being the most likely number. (*Could* is used in the preceding sentence because many scientists feel that as a result of the efficiency of the body's repair mechanisms at the very low dose rates involved, the true effects might approach zero production of cancer.)

The above numerical values are in essential agreement with those reported by the ICRP [3] and the UNSCEAR Committee [4]. The latter report stresses that the risk estimates are valid only for the doses at which they have been estimated (high levels), whereas the BEIR report suggests that the values are useful as upper-limit estimates in assessment of effects at low levels. A recent NCRP report [6] discusses this matter critically and concludes that the BEIR values have such a high probability of overestimating the actual risk that they are of only marginal value, if any, for purposes of realistic risk-benefit evaluation. At this time, we judge the consensus to be that the BEIR values are most likely overestimated by a considerable margin, but if used with that understanding, then there are important comparisons that can be made.

One shortcoming of these estimates is that they leave unanswered the question of the latent period between exposure and development of cancer, or to put that same issue differently, the amount of life shortening that will result. There is some evidence that latency may differ for cancer of different organ systems and may also differ for different conditions of exposure. For example, the latency prior to the development of leukemia was shorter for those persons treated radiotherapeutically to the spine for arthritis than for atomic bomb survivors.

6. Assessment of Risk from Nuclear Power

Health risks to both the general public and occupational personnel from the nuclear fuel cycle are considerably better estimated than those from fossil fuel combustion. This is because (a) there is a single causative agent released from the nuclear plant--ionizing radiation--whereas there are literally hundreds of individual species released from fossil fuel combustion; (b) since the first nuclear weapons tests in the 1940s, about a billion dollars have been spent on research on the effects of ionizing radiation; (c) radiation exposures are easily and precisely measured; and (d) there is a great body of knowledge from natural background exposure and from accidental, industrial, and military exposures of populations.

Table 1 shows the UNSCEAR estimates of man rem exposures to the public, both locally and worldwide per MW(e) per year. Typically, a roughly equivalent total exposures will occur to occupational personnel, but are not similarly distributed in the same portions of the fuel cycle. A large source of occupational exposure occurs to the underground uranium miner whose lungs are exposed to the radioactive gas, radon, and to the alpha particles of the radon "daughter products" which may absorb to ambient chest particles and deposit in the bronchi of the miners lung. Lung cancer has been demonstrated in this group, particularly in those who are also cigarette smokers.

Utilizing a "value of life" approach and the radiation risk estimate described above, a dollar cost per man rem can be calculated. For example, assigning a 300,000 dollar value to life and using the risk value of 100 cases per million man rem, simple arithmetic produces a 30 dollar per man rem value.

In addition to risks from radiation exposures discussed above, some specific issues and concerns have been raised which will be touched on briefly here.

6.1 Accidents

The basic document in regard to reactor accidents is the "Rasmussen Report" [8]. It attempts to predict the probabilities and consequences of a total spectrum of conceivable reactor acci-

Table 1: Summary of Collective Doses per Unit Energy Generated*

	Whole body [man rad/MW(e)y]				Referred organ [man rad/MW(e)y]				
A. LOCAL AND REGIONAL CONTRIBUTIONS									
Mining and milling	$3\ 10^{-3}$				Lung	$5\ 10^{-1}$			
Fuel fabrication	$1\ 10^{-4}$								
Reactor operation	PWR	BWR	GCR	HWR		PWR	BWR	GCR	HWR
1. Atmospheric pathways	$1\ 10^{-2}$	$5\ 10^{-1}$	$2\ 10^{-1}$	$6\ 10^{-2}$	Thyroid	$5\ 10^{-3}$	$5\ 10^{-1}$	$5\ 10^{-3}$	$5\ 10^{-3}$
2. Water pathways	$2\ 10^{-2}$	$3\ 10^{-2}$	$2\ 10^{-2}$	$3\ 10^{-2}$	Thyroid	$1\ 10^{-4}$	$1\ 10^{-2}$	$1\ 10^{-4}$	$1\ 10^{-4}$
Fuel reprocessing									
1. Atmospheric pathways	$5\ 10^{-3}$				Thyroid	$3\ 10^{-3}$			
2. Water pathways	$2\ 10^{-2}$				Thyroid	$6\ 10^{-2}$			
Transportation	$3\ 10^{-3}$								
B. GLOBAL CONTRIBUTIONS									
^{3}H	$1\ 10^{-1}$								
^{85}Kr	$2\ 10^{-1}$								
^{14}C	2.5								
^{129}I					Thyroid	$2\ 10^{-2}$			

*From [4].

dents. Critical reviews of this report have been made by the American Physical Society [9] and the Union of Concerned Scientists [10]. The essence of these analyses is that the Rasmussen estimates would have to be low by 3-5 orders of magnitude in order for the risks from catastrophic accidents to be comparable to those from normal operations of the coal, oil or nuclear fuel cycles. It is a matter of conjecture whether the public would accept the probability, although very small, of a single nuclear event causing an immediate loss of hundreds of lives as preferable to or in place of the loss of a large number of lives from fossil combustion occurring in driblets and therefore unnoticed.

6.2 Plutonium

Of all the radionuclides involved in the nuclear fuel cycle, plutonium has aroused the greatest public concern in regard to potential hazard. A great deal of experimental work has been done over the years on the biological effects of plutonium [11, 12]; but of course as with other toxic substances it is not possible to predict precisely the effects of low levels in the range of exposure that would produce undetectable effects.

Following is a discussion of those factors that tend to cause plutonium-239 to be hazardous, and then of those that tend to reduce its hazard. Plutonium, as any alpha-emitting radionuclide, is very biologically effective in producing cancer when it is located within the body in direct contact with living tissues. When it is inhaled it comes into direct contact with living tissue, and when it enters the blood it is deposited in such tissues as bone, liver, and lymph node; once deposited it remains for a long time during which it irradiates the tissue. Because of its long physical half-life (24,300 years) it must be regarded essentially as a permanent contaminant just as are many other stable idustrial chemicals that pollute the biosphere.

Because alpha radiation will not even penetrate the dead layer of skin, plutonium is not a hazard when it exists outside of the body. Contrary to popular conception, plutonium when swallowed remains essentially outside the body because it is extremely poorly absorbed, does not enter the bloodstream in significant proportions, and being mixed with intestinal contents does not irradiate the surface of the intestines as it passes through the gastrointestinal tract. Plutonium does not become concentrated in the food chain. These characteristics result in large part from the low solubility of plutonium in water and biological fluids and its tendency to remain fixed in soil.

It appears that inhalation of plutonium is the most hazardous route of exposure. Because plutonium deposited in the lung may be presented as small particles, a question has been raised as to whether a given amount of such radioactivity deposited in the lung would be more hazardous if present as small particles rather then being uniformly deposited. This is presently a matter of controversy. One group of workers [13] claims on the basis of theoretical considerations that small particles would be more hazardous (hot particle theory) and therefore that existing standards, which are based on uniform distribution, should be made more stringent. Other workers and several official groups claim that experimental data support existing concepts, and that there is no reason for any drastic change of standards [14-17].

The problem of malevolent use of plutonium cannot be logically assessed; this matter has been discussed by Cohen [18]. It appears that except for an unreasoning widespread public fear, terroristic purposes could be much more readily achieved by using other more easily available chemical or biological agents.

In general it can be stated that plutonium when inhaled is a toxic carcinogen and great care should be taken to prevent its access to the biosphere. Essentially none would be released from normal operation of the nuclear fuel cycle. Estimates of risks from it as a component of nuclear fuel cycles and the experience of the past 30 years indicate that they are lower than from other parts of the cycle and from other fuel systems.

7. Fossil Fuels

The data presented on the health effects associated with fossil fuels suffer from certain limitations and uncertainties. First, genetic effects are not included because our present state of knowledge does not allow even an approximate estimate for such effects. Second, the data do not adequately discriminate between premature deaths that may occur early in life, such as from accidents, and those that may shorten life only slightly, as seen, for example, in increased mortality among persons hospitalized for chronic disease, who already have high mortality rates. Perhaps of greatest importance is the uncertainty about the validity of the upper estimates for the effects on the general public from burning coal and oil. Not only is there the problem of the magnitude of the effect, but lack of knowledge about the causative agents makes it difficult to institute effective control procedures.

The primary data come from epidemiological studies. Major episodes (Meuse, Donora, London, New York City, etc.) clearly showed that air pollution, sufficiently severe, could cause illness and premature death. During the 1950s and 1960s the major issue was whether air pollution in concentrations usually existing over industrial cities would cause adverse health effects. The emphasis shifted next to quantifying pollution relative to effects produced and more recently to the effects of low levels of pollution and the effects of interactions.

From a methodological standpoint epidemiological, animal, and experimental human studies are needed. Epidemiological studies are important in uncovering possible associations that can then be tested under controlled conditions; they are also needed for evaluation of human risks suggested by laboratory experiments. Animal studies are used to determine efficiently the sites of effects, mechanisms, and dose--response relationships, and they are more easily adapted for chronic studies than are human investigations. Because of species differences, controlled studies on humans are needed to establish responses and to determine the influence of disease or of various physiological states on the effects of pollution.

Since all of the human data from which damage functions have been drawn rest on the use of regression analysis, we offer in the Annex some brief comments on that technique so that the reader may be aware of the use and limitations of the method.

8. Air Pollution Damage Functions

In calculating the health effects of energy, it is sometimes possible to rely on actual experience. For example, reliable records of accidents incurred in coal mining exist [21]. These can be used to calculate the expected fatality or injury rate per unit of coal extracted. Other effects, such as radiation hazards, have been statistically estimated from a wealth of data as discussed above. The effects of chemical air pollution, however, have been more elusive. The derivation of quantitative damage functions has been attempted only in the past decade and researchers relied almost exclusively on regression analysis. Furthermore, since reliable morbidity data are difficult to obtain, most studies have restricted themselves to mortality rates. The major studies in this area are summarized here.

Lave and Seskin [22] considered 117 standard metropolitan statistical areas (SMSAs) of the U.S. As dependent variables, they used total, infant, and certain disease-specific mortality rates. The independent variables included the percent non-white population, the population density, the percent of population over 64 years old, and the percent poor population, i.e. families with annual income under $ 3000. The air pollution data consisted of 26 biweekly concentration measurements of suspended particulates and total sulfates. From these, they used as independent variables the minimum, maximum and mean of each group of 26 values. The data were collected for the years 1960 and 1961 and multiple linear regression were performed for each year. The following is an example of their 1960 regression results:

Total mortality rate (per 10,000 people) = 19.607
+ (0.041) mean suspended particulates ($\eta g/m^3$)
+ (0.71) minimum sulfate ($\eta g/m^3$)
+ (0.001) number of persons per square mile
+ (0.0041) percent non-white
+ (0.0687) percent over 65 years.

In this equation, only the variables whose regression coefficients were statistically significant at the 0.05 level were included. To understand the relative practical significance of each variable, the authors calculated the elasticity. That is, based on the regression coefficients, they calculated the percent increase in the mean value of mortality rate if the mean of each of the independent variables was increased by 10%. These calculations are presented in Table 2. Thus, understandably, mortality rates are most sensitive to the older segment of the population. Lave and Seskin conclude from these elasticities that mortality would decrease by 4.5% if the level of pollution was decreased by 50%.

Table 2: Elasticities of Air Pollution and Socio-economic Variables*

Variable	Increase in Mortality Due to 10% Increase
Mean P	0.53%
Min S	0.37%
P/M^2	0.07%
% N-W	0.57%
% 65	6.32%

*From [22].

In another paper, Lave and Seskin [23] considered the effects of some meteorological and home heating variables. The pollution variables remained statistically significant and the magnitude of the regression coefficients did not change appreciably. The authors also conclude that transformations of the variables did not improve the goodness of fit [24].

Winkelstein et al. [25] studied mortality rates from all causes among white males 50-69 years of age in Erie County, New York. His statistical unit was the census tract (125 in all). He tabulated these death rates for subgroups of this population, based on the socio-economic levels and mean air pollution measurements. The strong associations suggested by these data prompted Hamilton and Morris [26] to compute the following regression equation:

$$\begin{aligned}&\text{Mortality rate of men 50-69 years per 1000}\\ &\quad = 33.97\\ &\quad + (0.15)\ \text{mean total suspended particulates } (\eta g/m^3)\\ &\quad - (0.0034)\ \text{mean family income } (\$).\end{aligned}$$

Hickey et al. [27] considered 15 measurements of atmospheric chemicals and mortality rates due to cancer and heart disease. This restricted their sample size to 38 SMSAs. The data were averages for the period 1957-1964. With no adjustments for age and socio-economic status, they obtained regression equations of these disease-specific mortality rates on the logarithms of the pollutant concentrations. The concentrations of SO_2 and NO_2 appeared consistently as significant predictors in these equations.

Carnow and Meier [28] used benzo[a]pyrene as an index of air pollution. Their dependent variables were age-specific death rates due to pulmonary cancer. They compared urban with rural, migrant with nonmigrant and smoking with nonsmoking populations. The independent variables were average cigarette smoking levels and benzo[a]pyrene concentrations in the 48 contiguous states of the U.S. For 19 highly developed countries they calculated regression equations with tobacco sales and consumption of solid fuels as independent variables. They summarized their study with the statement: "A reduction of 60% in urban air pollution might be expected to reduce the deaths from pulmonary cancer by 20% in all smoking categories."

Schwing and McDonald [29] noted that these and other regression studies suffered from two limitations:

(1) They included a limited number of pollution measurements; and

(2) They used ordinary least squares to estimate the regression coefficients.

In attempting to overcome these limitations, they included "a rather broad (but still incomplete) list of explanatory variables". These consisted of seven chemical pollution measurements, two radiation values, tobacco sales, four weather variables and nine variables describing population and socio-economic distributions. The dependent variables were mortality rates among white males for the 15 leading causes of death, age-stratified total deaths and age-stratified deaths due to lung cancer and heart disease. The sample consisted of 46 SMSAs. In all, they computed 40 regression equations, one for each of the disease and age categories. The authors also calculated elasticities for the pollution, radiation and smoking variables. Although the results were not always consistent, concentration of sulfur compounds and cigarette smoking were generally strongly associated with mortality. Associations with nitrogen compounds, the hydrogen index used, and ionizing radiation were less conclusive.

Later in this paper, we present a summary of calculations of health effects of energy based on these and other damage functions. To illustrate the difference between these functions, we present calculations made by Hamilton and Morris [26] based on the above mentioned equations derived by Winkelstein (W) [25], Schwing and McDonald (S-MC) [29] and Laven and Seskin (L-S) [22]. They considered a 1000 MW(e) coal-fired power plant with a 1000 foot stack, using 3% sulfur coal, 12% ash and 99% particulate removal. In an 80 km radius area with "typical" population distribution (164,000 people), they calculated the expected ambient concentrations of SO_2 and sulfur particulates. Applying the above equation, Table 3 of expected "excess deaths" was generated.

Table 3: Comparison of "Excess Mortality" Based on Various Damage Functions*

Damage Function	Excess Mortality
Men 50-69 (W)	29.0
Men 50-69 (S-MC)	0.1
Total male (S-MC)	90.0
Total population (L-S)	19.0

*From [26].

It is interesting to note that these estimates are within two orders of magnitude of each other. This is typical of the calculations made by different researcher in this area and reflect the state of our knowledge at the present time.

In 1970, air quality standards for selected pollutants were mandated by the United States Clear Air Amendments. Emphasis was placed on sulfur dioxide because of the evidence that ambient levels were associated with health effects of air pollution disasters. Subsequent studies indicated that sulfur dioxide by itself could not be the primary causative agent and it was postulated that a combination of sulfur dioxide and particulates was responsible [30-32]. More recent evidence suggests that oxidation products of sulfur dioxide (i.e. sulfuric acid and particulate sulfates)--possibly acting synergistically with sulfur dioxide and other pollutants such as nitrates, particles, and ozone--are primarily the causative agents [33-35]. It must be emphasized that although suspended sulfates are now being used as an indicator of health effects and there appear to be correlations between them and such effects, there is no firm evidence as to which substance or substances in polluted air are the causative agents. Without such knowledge, air pollution control strategy based on reduction of sulfur alone does not have a valid scientific basis.

The major categories of health effects associated with air pollution are (a) chronic respiratory disease; (b) symptoms of aggravated heart-lung disease; (c) asthma attacks; (d) children's respiratory disease; and (e) premature death. It would be most useful to understand the quantitative relationships between exposures to specific agents and these health effects in order to know how much investment is justified for control measures, to know which chemical effluents to control, and to make comparisons with biological costs of nuclear power.

In a recent report of the National Academy of Sciences-National Academy of Engineering-National Research Council [36], illustrative calculations were made of the health effects associated with sulfur oxide emissions for representative power plants in the Northeast. The results are presented in Table 4. They were derived from models that related ambient levels to emissions including factors for conversion of SO_2 to sulfates; health effects from ambient levels were calculated by using dose-response curves from epidemiological data from studies of the Environmental Protection Agency (EPA). It must be emphasized that the numerical estimates of Table 4 are controversial, relying on limited information and numerous arbitrary assumptions, and cannot be regarded as proven results. A critique in the same document from which Table 4 was derived [36, Chapter 4] suggests that the estimates could be low by a factor of two or high by a factor of ten. What can be concluded from Table 4 with reasonable assurance is that the effects listed are produced at detectable levels by factors associated with air pollution, with power plants most likely making a significant contribution. It should also be noted that a cost-benefit assessment of the data in Table 4 indicates that the economic impact of the nonlethal effects is much greater than of the premature deaths.

Table 4: Health Effects Associated with Sulfur Oxide Emissions*

	Remote Location	Urban Location
Case of chronic respiratory disease	25,600	75,000
Person-days of aggravated heart-lung disease symptoms	265,000	755,000
Asthma attacks	53,000	156,000
Cases of children's respiratory disease	6,200	18,400
Premature deaths	14	42

*Source: [36, Chapter 13]. Illustrative calculations based on distributive models, postulated conversions of SO_2 to SO_4, and EPA epidemiological data for representative power plants in the Northeast emitting $96.5 \cdot 10^6$ pounds of sulfur per year--equivalent to a 620 MW(e) plant.

9. Health Effects from Electricity Generation

Several reports have been published that contain estimates of the health effects associated with electricity production [7, 21, 37-42]. Tables 5 and 6 summarize the available estimates for each phase of the fuel cycle for each of the four fuels: coal, oil, natural gas, and nuclear. By and large, the estimates relate to contemporary technology and existing circumstances. In each case the data have been adjusted to represent the number of premature deaths or occupational impairments produced per year by processes associated with a 1000 MW(e) power plant, which is roughly that required for a population of 1,000,000 people. The values given represent the lowest and highest from the cited references. The references should be consulted for an understanding of the methodology and detailed assumptions; limitations have been discussed in the previous section.

Consider first from Table 5 the effects on workers. For coal-fired plants the values range from 0.5 premature deaths per year and for the other fuel sources they range somewhat lower, from 0.06 to 1.3. Most of these effects are due to accidents in coal mines, to conditions that cause black lung disease, or to activities in oil refineries, uranium mining, and nuclear fuel reprocessing.

Consider now from Table 5 the effects on the general population. It has been estimated that the transport of coal required for a year's operation of a 1000 MW(e) plant is responsible for 0.6 to 1.3 premature deaths by accidents at railroad crossings; no estimates are available for truck or barge transport. The comparative values for the other fuel systems are insignificant.

The data so far discussed have a reasonable statistical base of past operation and are to that extent reliable. The number of premature deaths among the public from power plant operation (conversion or generation of electricity) results primarily from dissemination of air pollutants and, as discussed earlier, these effects are a matter of great uncertainty. The upper-limit estimates for coal and oil are about 100 premature deaths per year compared with 1 or less for natural gas and nuclear.

Table 6 presents data on the number of nonfatal occupational injuries per year associated with the operation of a 1000 MW(e) power plant. These have been defined as injuries serious enough to cause loss of working time for several days or more. These effects are roughly the same for coal and oil, ranging from about 12 to 100 cases per year, and somewhat lower for natural gas and nuclear. Most of these effects are associated with mining, well digging, coal transport, oil refining, and nuclear reprocessing.

Table 5: Premature Deaths per Year Associated with Operation of a 1000 MW(e) Power Plant (values are lowest and highest estimates from cited references)[a]

	Coal	Oil	Natural Gas	Nuclear
Occupational				
Extraction				
Accident	0.45-0.99[21,37,39,40,42]	0.06-0.21[37-40,42]	0.021-0.21[37-40,42]	0.05-0.2[7,37,39,40,42]
Disease	0-3.5 [39]	-	-	0.002-0.1[7,39,41,42]
Transport				
Accident	0.055-0.4 [37,39,40,42]	0.03-0.1[37-39,42]	0.02-0.024[37,39,40,42]	0.002[37,40,42]
Processing				
Accident	0.02-0.04[39,40]	0.04-1[37-40,42]	0.006-0.01[37,39,40,42]	0.003-0.2[7,37,39,40,42]
Disease	-	-	-	0.013-0.33[7,39,41,42]
Conversion				
Accident	0.01-0.03[37-40,42]	0.01-0.037[37-40,42]	0.01-0.037[37-40,42]	0.01[37,39,40,42]
Disease	-	-	-	0.024[7]
Subtotals				
Accident	0.54-1.5	0.14-1.3	0.057-C.28	0.065-0.41
Disease	0-3.5	-	-	0.039-0.45
Total	0.54-5.0	0.14-1.3	0.057-0.28	0.10-0.86
General Public				
Transport	0.55-1.3[21,37,39,42]	-	-	-
Processing	1-10[39]	-	-	-
Conversion	0.067-100[21,39]	1-100[39]	-	0.01-0.16[b][7,37,39,41,42]
Total	1.6-111	1-100	-	0.01-0.16
Total Occupational and Public	2-116	1.1-101	0.057-0.28	0.11-1.0

[a]Note: Dashes indicate no data found; effects, if any, are presumably too low to be observed; and no theoreticl basis for prediction. From [43].

[b]For processing and conversion.

Table 6: Occupational Injuries per Year Associated with Operation of a 1000 MW(e) Power Plant (values are lowest and highest from cited references)*

Occupational Injuries	Coal	Oil	Natural Gas	Nuclear
Extraction				
Accident	22-49[37,39,40,42]	7.5-21[37-40,42]	2.5-21[37-40,42]	1.8-10.0[37,39,40,42]
Disease	0.6-48[21,39]	-	-	-
Transport				
Accident	0.33-23[37,39,40,42]	1.1-9[37-39,42]	1.2-1.3[37-39,42]	0.045-0.14[37,40-42]
Processing				
Accident	2.6-3[39,40]	3-62[37-40,42]	0.05-0.56[37-39,42]	0.6-1.5[37,39,40,42]
Conversion				
Accident	0.9-1.5[37-40,42]	0.6-1.5[37-40,42]	0.6-1.5[37-40,42]	1.3[37,39,40,42]
Totals				
Accident	26-77	12-94	4-24	4-13
Disease	0.6-48	-	-	-

*From [43].

10. Areas in Need of Further Research

Several issues and controversies regarding health effects of energy need to be settled. On the biological side, the mechanisms involved in the effects of radiation are poorly understood and those of chemical pollution even more so. This is reflected in the controversy of threshold versus linear extrapolation theories alluded to earlier. Further light may be shed on the problem if scenarios are constructed where each theory is adopted in turn. On the one hand (threshold), some deleterious effects on health may be neglected while on the other (linear) the use of energy may be needlessly restricted. Balancing these two types of "errors" at our present incomplete state of knowledge is a pragmatic issue which needs to be resolved.

Another pragmatic issue is to find a common index for various health effects which may be quantitatively and qualitatively different. On the one hand, chemical versus radiation effects and on the other normal operations versus accidental effects. This involves comparison of short-term somatic effects with long-term genetic effects. It also involves effects on which varying magnitudes of data are available and some on which no data are available (and hopefully never will be).

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Regression Analysis

Regression analysis is a statistical technique used to study the relationship between a criterion (or dependent) variable and a set of explanatory (or independent) variables. For example, the dependent variable may be mortality rate and the independent variables may be various measurements relating to air pollution, health conditions and socio-economic status of a given population.

Denoting the dependent variable by Y and the independent variables by $X_1,...,X_p$, we can postulate the functional relationship:

$$Y = f(X_1,...,X_p) \quad .$$

The major statistical problem is how to best estimate the parameters of this function. To this end, a number of measurements from different regions (cross-sectional) or at different time points (longitudinal) must be obtained. Then the parameters can be estimated from the data using, e.g. the least squares or maximum likelihood methods.

Some assumptions are usually made:

(i) independence, i.e. a data point does not affect, and is not affected by any other data point;

(ii) homoscedasticity, i.e. the variance of the distribution of Y at a given combination of values of $X_1,...,X_p$ is the same as that at any other combination.

Another assumption is often made in order to be able to test hypotheses about the parameters of the regression function f; namely:

(iii) normality, i.e. the distribution of Y at any combination of values of $X_1, \ldots, X_p$ is normal (or Gaussian).

To proceed with the analysis, the form of the function f must be specified. The one used in most applications is the linear function, i.e.

$$f(X_1, \ldots, X_p) = \alpha + \beta_1 X_1 + \ldots + \beta_p X_p \quad ,$$

where $\alpha, \beta_1, \ldots, \beta_p$ are unknown parameters to be estimated from the data. The main advantage of this function is its simplicity; Y changes by an amount β_1 if the value of X_1 is increased one unit (provided $X_2, \ldots, X_p$ are not changed). The estimates of the parameters are easily obtained by the method of least squares. Indeed, several efficient packaged computer programs exist for this purpose [19]. Furthermore, the least squares estimates are optimal under assumptions (i) and (ii) [20].

The disadvantage of the linear model is that it may not provide an adequate description of the underlying functional relationship. If a nonlinear regression function is assumed, however, complex iterative procedures must be used to estimate the parameters. When the number of independent variables is large, the calculations become difficult, even with the aid of a large computer. This leads many researchers to using linear functions as approximations. The degree of approximation can be improved by limiting the range of variables, limiting the diversity (or span) of the population, or making transformations of the variables.

Although the regression curve may be non-linear over a wide range of the variables, it is frequently possible to consider only limited ranges over which the curve can be approximated reasonably well by a straight line. Similarly, if the population (or time span) under study is restricted, a linear regression may prove an adequate model. Finally, transformation of some or all the variables can produce (at least approximately) linearity. For example, suppose that the regression of Y on one independent variable X is

$$Y = e^{\alpha+\beta x} \quad .$$

Then,

$$\log Y = \alpha + \beta x \quad ,$$

which is a linear regression equation of log Y on X. Quadratic or higher order powers of the independent variables can also provide an approximation of the non-linear regression curve while keeping the model linear in the parameters.

At this point some words of caution about interpreting the results of regression analysis may be useful. In using data not collected from a planned experiment, it is rarely possible to control for, or include measurements on all of the factors involved. Therefore, based on the regression results alone, it is not possible to infer causal relationship. The variables measured often act as surrogates of the underlying, unmeasured, causal factors. Furthermore, intercorrelations, i.e. collinearities among the independent variables often make it difficult to quantify the effect of an individual independent variable on the dependent variable. Thus, although regression analysis is a powerful predictive tool, it must be used only with caution as a normative explanatory technique.