Modelling Standard Setting Decisions: An Illustrative Application to Chronic Oil Discharges

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MODELLING STANDARD SETTING DECISIONS: AN ILLUSTRATIVE APPLICATION TO CHRONIC OIL DISCHARGES

BY

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Preface

Standard setting is one of the most commonly used regulatory tools to limit detrimental effects of technologies on human health, safety, and psychological well being. Standards also work as a major constraint on technological development, particularly in the energy field. The trade-offs which have to be made between economical, engineering, environmental, and political objectives, the high uncertainty about environmental effects, and the conflicting interests of groups involved in standard setting, make the regulatory task exceedingly difficult.

Realizing this difficulty, the Volkswagen Foundation sponsored a research subtask in IIASA's Energy Program under the name "Procedures for the Establishment of Standards". The objectives of this research are to analyze existing procedures for standard setting and to develop new techniques to improve the regulatory decision making process. The research performed under this project include:

i) policy analyses of the institutional aspects of standard setting and comparisons with other regulatory tools;

ii) case studies of ongoing or past standard setting processes (e.g., oil discharge standards or noise standards);

iii) development of formal methods for standard setting based on game and decision theory;

iv) applications of these methods to real world standard setting problems.

The present research memorandum is one in a series of papers dealing with the development and application of decision theoretic models for standard setting. It presents an illustrative application of a model developed at IIASA to the problem of setting chronic oil discharge standards.

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.

<u>Contents</u>

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 ecclogical 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 $a, b \in A$;
2. A set of possible events S with may influence the consequence of an action a, with typical elements $s, t \in S$;
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 $C = \times_{i=1}^{n} C_i$ 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 $s \in S$ 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:

$$U(a) = \int_C \left[\sum_{s \in S} p(s|a) \cdot f(\underline{c}|a,s) \right] \cdot h[v(c)] \, d\underline{c} \tag{1}$$

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

$$U(a) = \sum_{s \in S} p(s|a) \cdot (\text{sign } \alpha) \exp\{\alpha \sum_{i=1}^{n} w_i v_i [c_i [a,s)]\} \tag{2}$$

for $\alpha \neq 0$ and

$$U(a) = \sum_{s \in S} p(s|a) \sum_{i=1}^{n} w_i v_i [c_i (a,s)] \tag{3}$$

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.

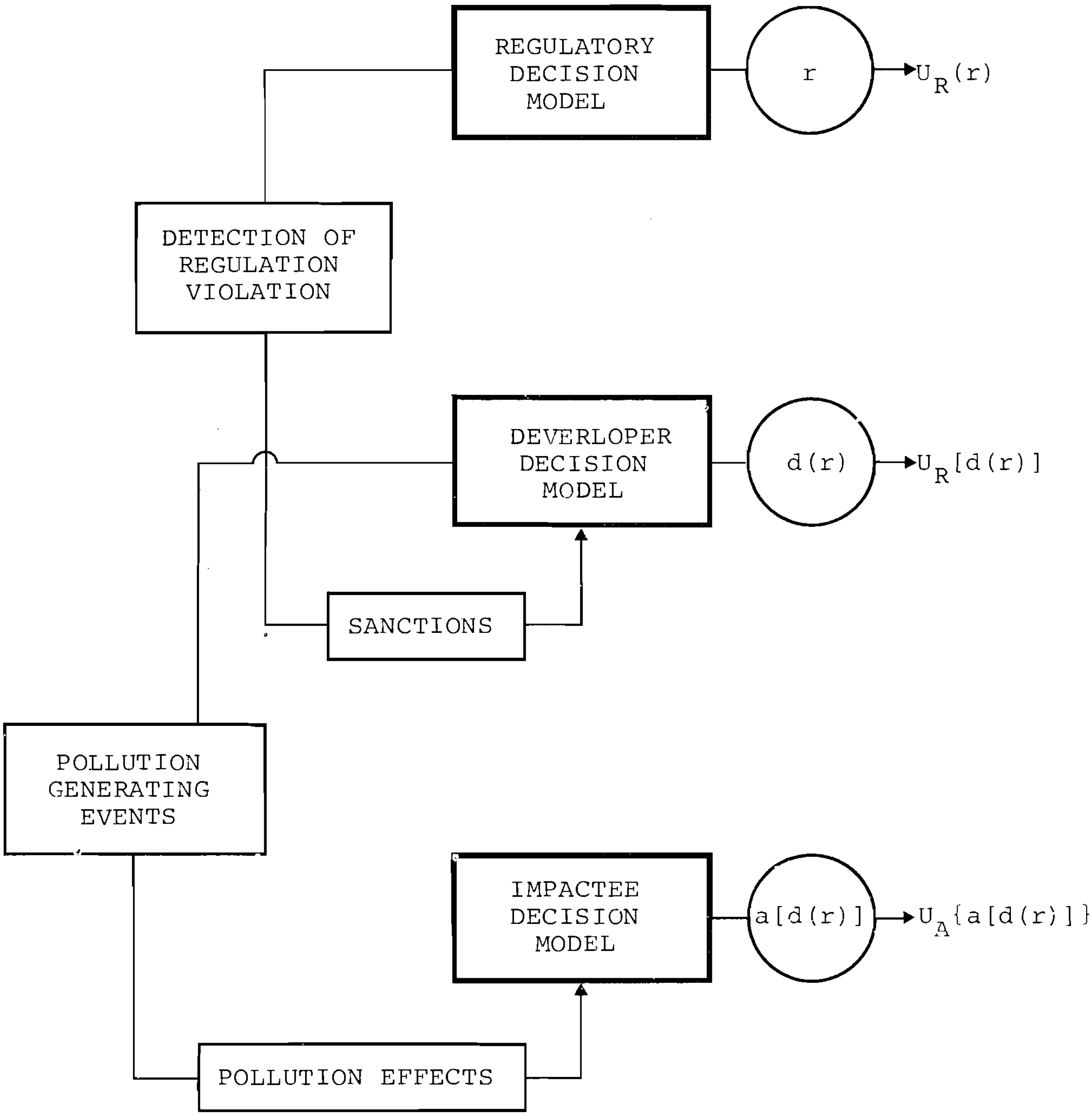

Figure 1 <u>SCHEMATIC REPRESENTATION OF THE REGULATOR-DEVELOPER-IMPACTEE MODEL</u>

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 $p_\cdot$ and $v_\cdot$ 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:

$$U_R(r) \quad = \sum_{i=1}^{n_R} w_{Ri} v_{Ri}[c_{Ri}(r)] \tag{4}$$

$$U_D(d,r) = \sum_{k=0}^{1} p_D(Q_k) \text{ (sign } \alpha) \exp\{\alpha \sum_{i=1}^{n_D} w_{Di} v_{Di}[c_{Di}(d,ss,Q_k)]\} \tag{5}$$

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:

$$v_{R1}(30) = 100 \tag{11}$$

$$v_{R1}(20) = v_{R1}(40) = 50 \tag{12}$$

$$v_{R1}(sl) \to 0 \quad sl \to \pm\infty . \tag{13}$$

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:

$$v_{R4}(40) = 100 \tag{23}$$

$$v_{R4}(20) = v_{R4}(60) = 50 \tag{24}$$

$$v_{R4}(sl) \rightarrow 0 \text{ for } sl \rightarrow \pm \infty . \tag{25}$$

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:

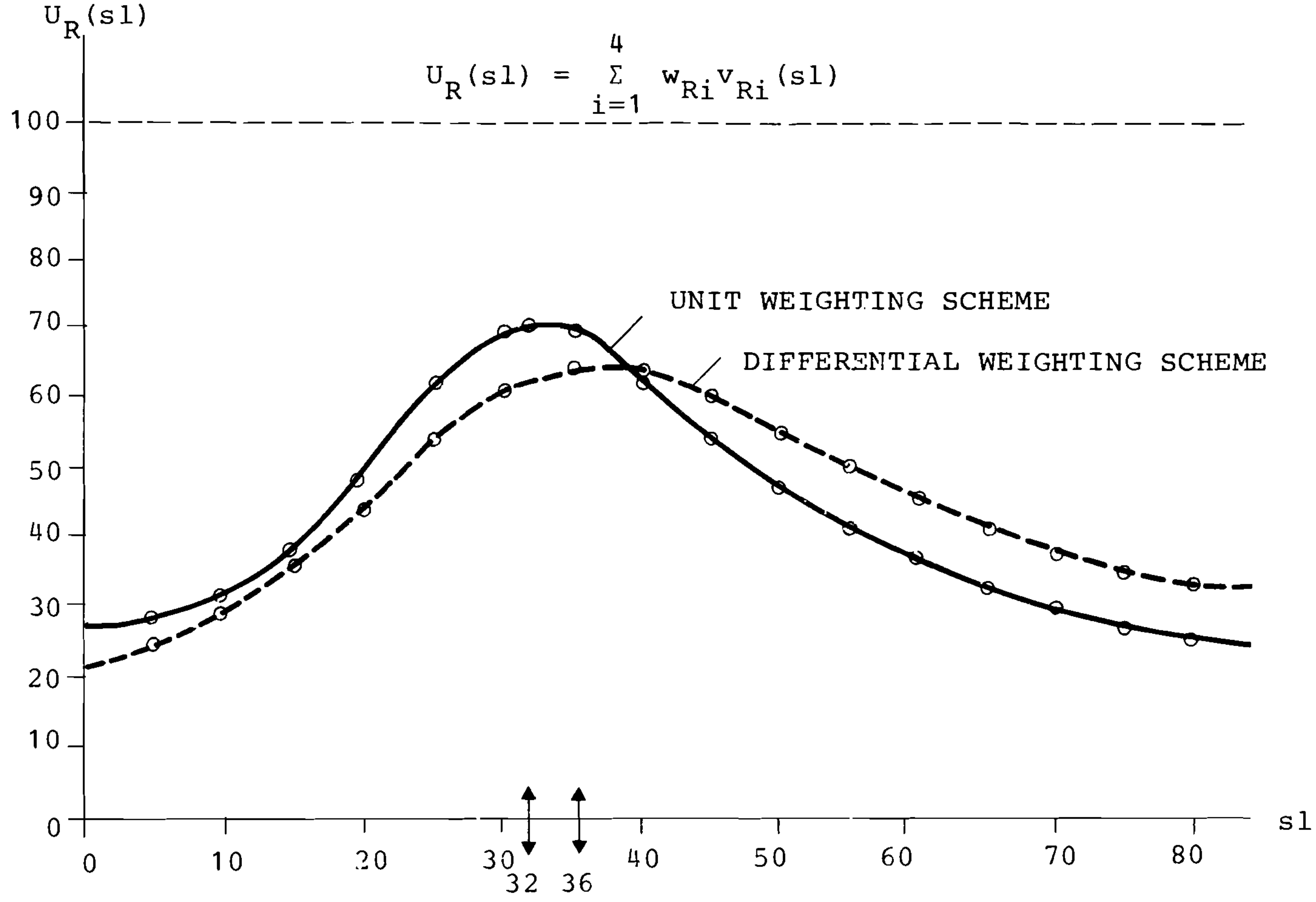

Figure 2 OVERALL VALUE FUNCTION OF THE REGULATOR FOR TWO WEIGHTING SCHEMES

(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:

$$\text{Prob}[\text{i violations}|p_o,M] = \binom{M}{i}(1-p_o)^i \, p_o^{M-i}. \tag{44}$$

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

$$p_D(Q_0|d_j,sl,sm) = p_m^N. \tag{47}$$

In EPA's monitoring and inspection example N would be 130.

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 "diluted" 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 $\hat{l}$ 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

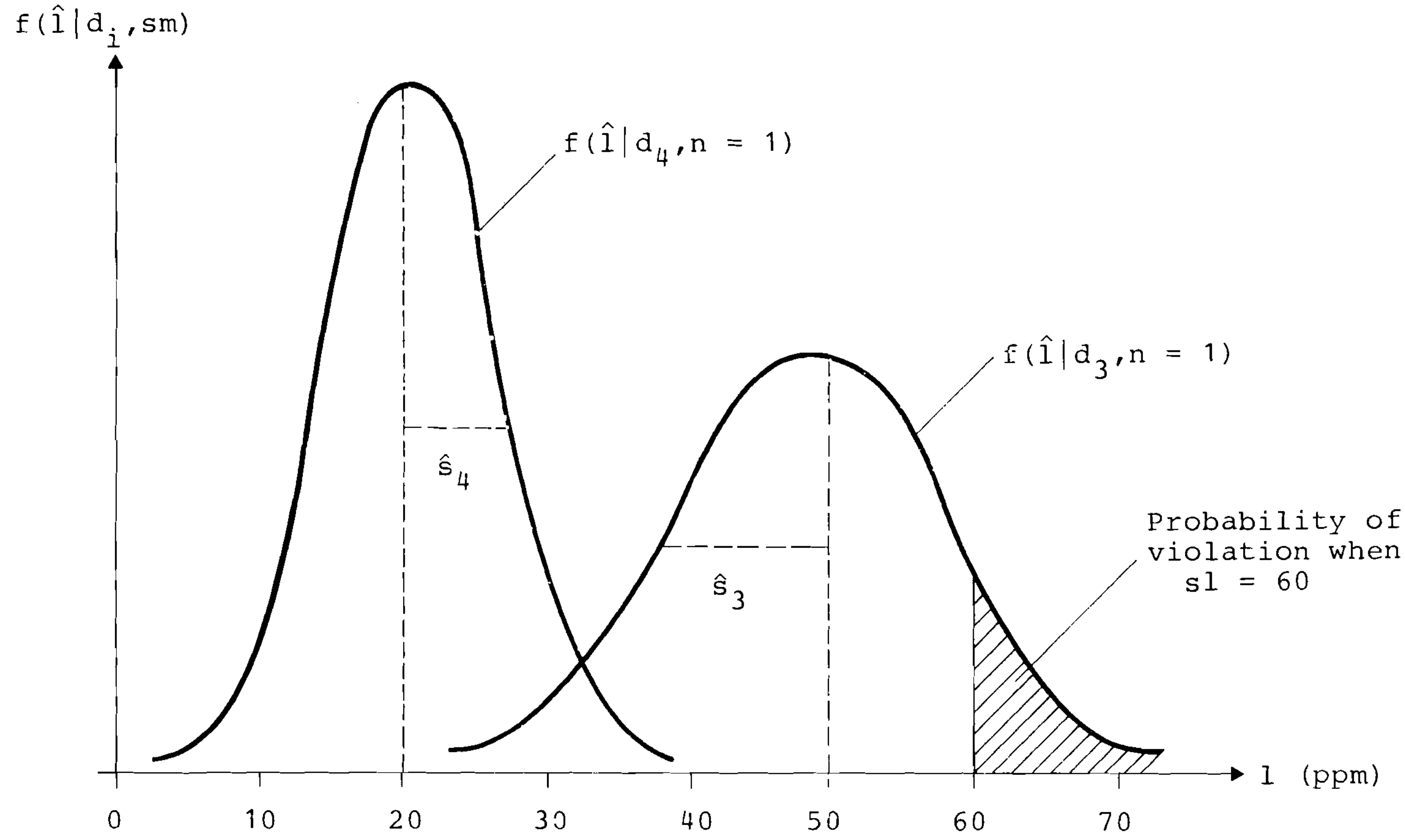

Figure 3 ILLUSTRATION OF TWO NORMAL DISTRIBUTIONS CHARACTERIZING TREATMENT PERFORMANCE UNCERTAINTY

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 ($\alpha = 2\cdot10^{-6}$)			RISK NEUTRAL UTILITY FUNCTION ($\alpha = 0$)			RISK AVERSE UTILITY FUNCTION ($\alpha = -2\cdot10^{-6}$)		
$K_0(10^3)$	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)

$K_0\,(10^3)$	$\alpha = 2 \cdot 10^{-6}$			$\alpha = 0$			$\alpha = -2 \cdot 10^{-6}$		
	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.

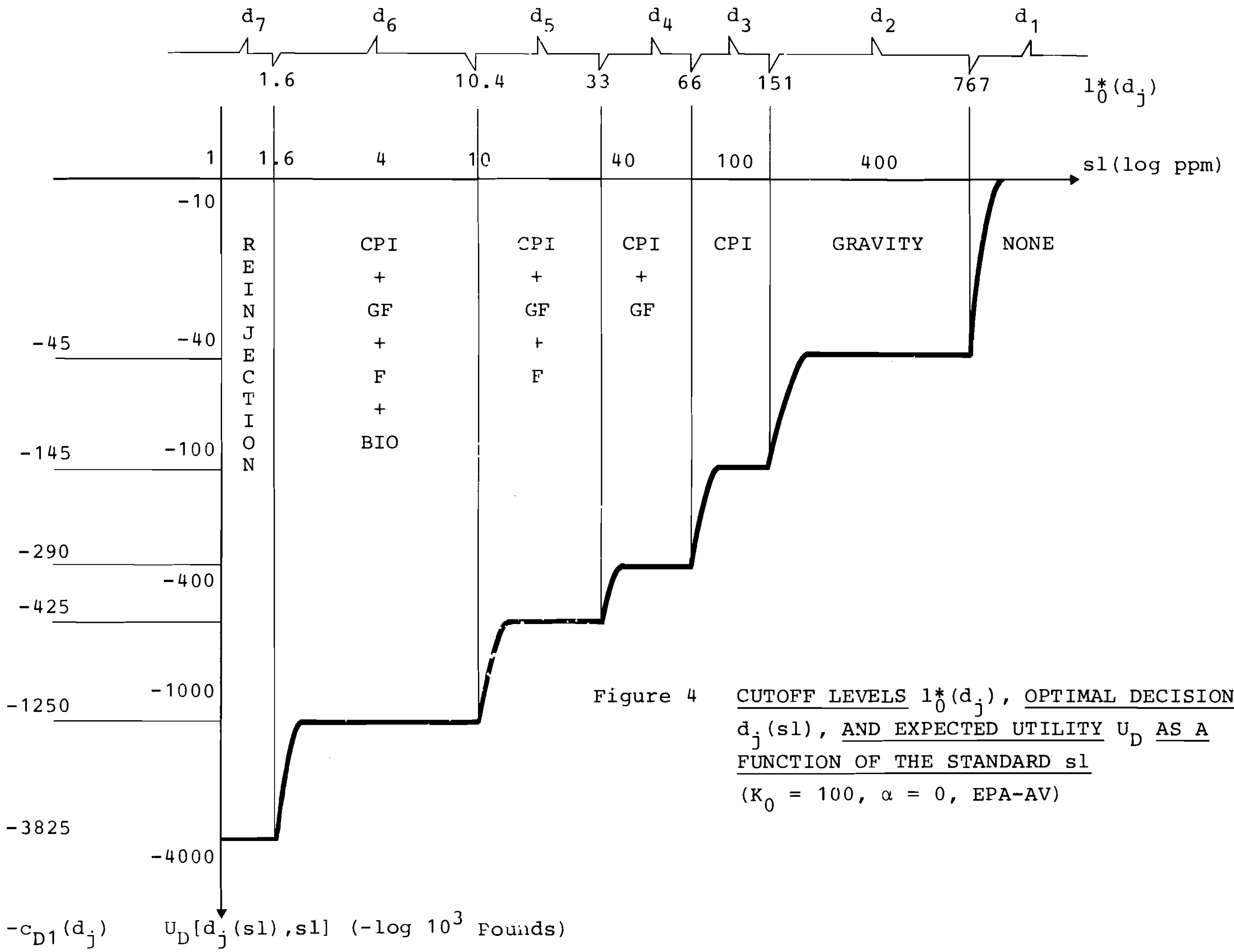

Figure 4 CUTOFF LEVELS $l_0^*(d_j)$, OPTIMAL DECISIONS $d_j(sl)$, AND EXPECTED UTILITY U_D AS A FUNCTION OF THE STANDARD sl
(K_0 = 100, α = 0, EPA-AV)

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., $s_j \rightarrow 0$ and $\bar{s}_j \rightarrow 0$). 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

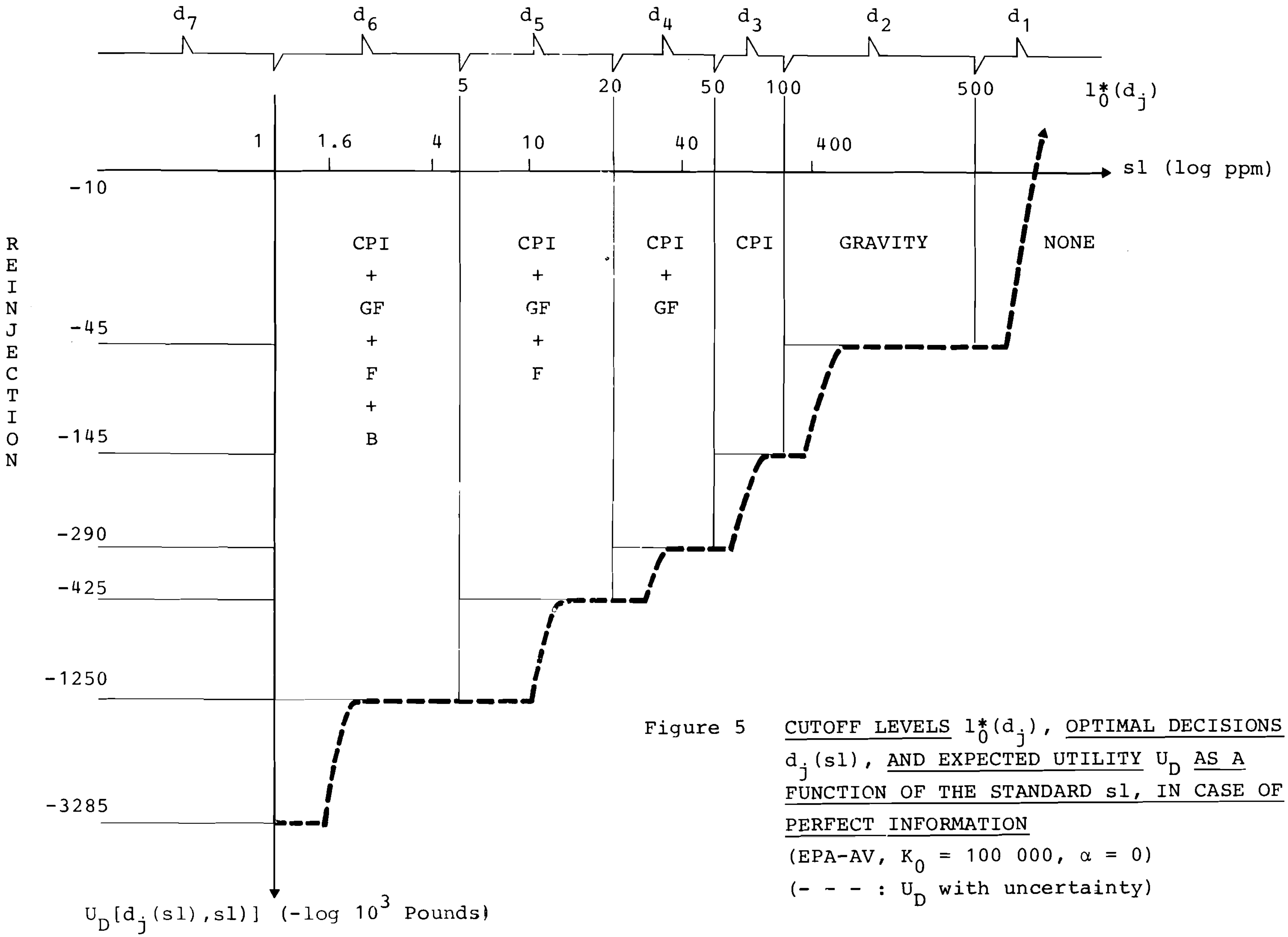

Figure 5 CUTOFF LEVELS $l_0^*(d_j)$, OPTIMAL DECISIONS $d_j(sl)$, AND EXPECTED UTILITY U_D AS A FUNCTION OF THE STANDARD sl, IN CASE OF PERFECT INFORMATION

(EPA-AV, K_0 = 100 000, α = 0)

(- - - : U_D with uncertainty)

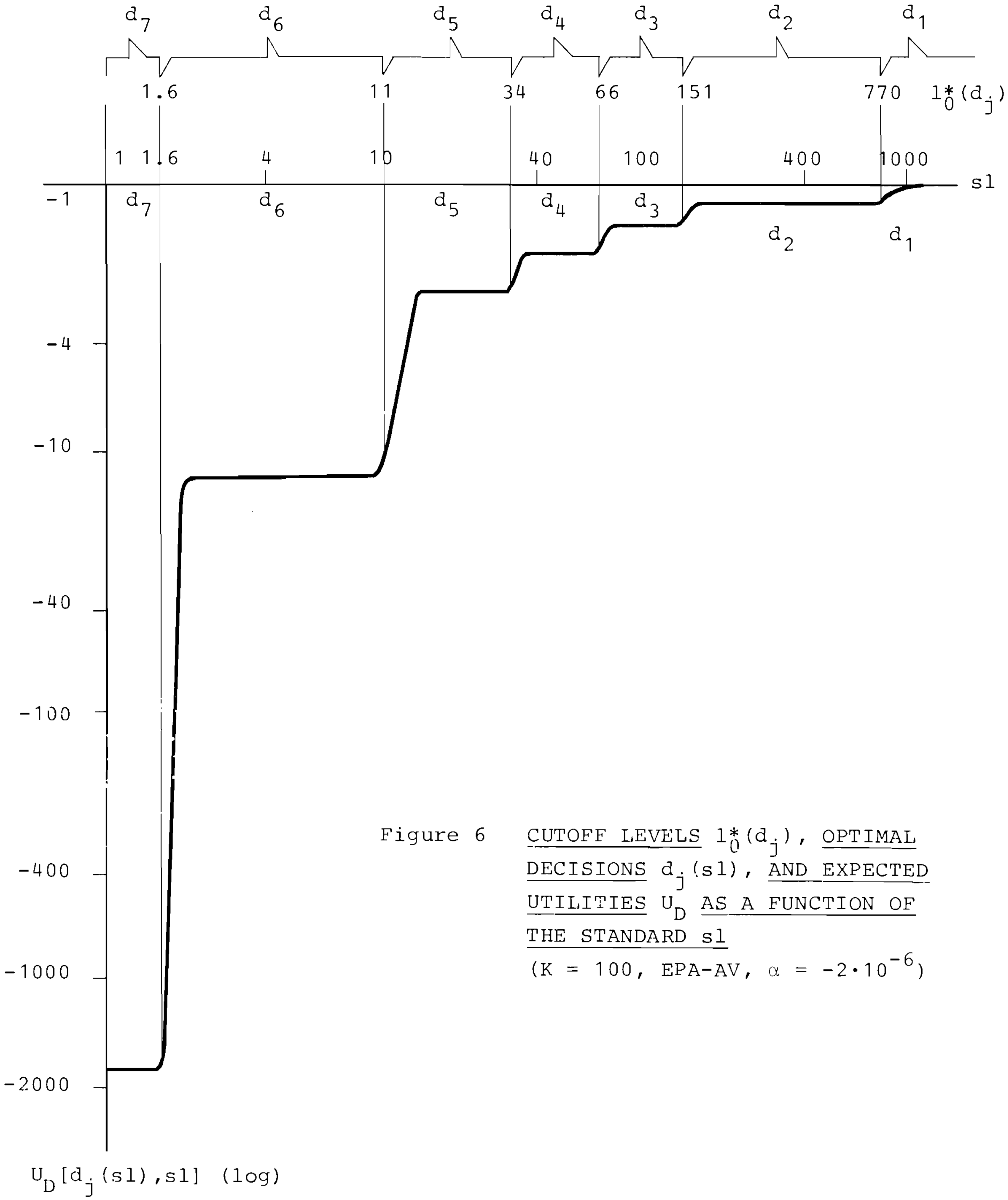

Figure 6 <u>CUTOFF LEVELS</u> $l_0^*(d_j)$, <u>OPTIMAL DECISIONS</u> $d_j(sl)$, <u>AND EXPECTED UTILITIES</u> U_D <u>AS A FUNCTION OF THE STANDARD sl</u>
(K = 100, EPA-AV, $\alpha = -2 \cdot 10^{-6}$)

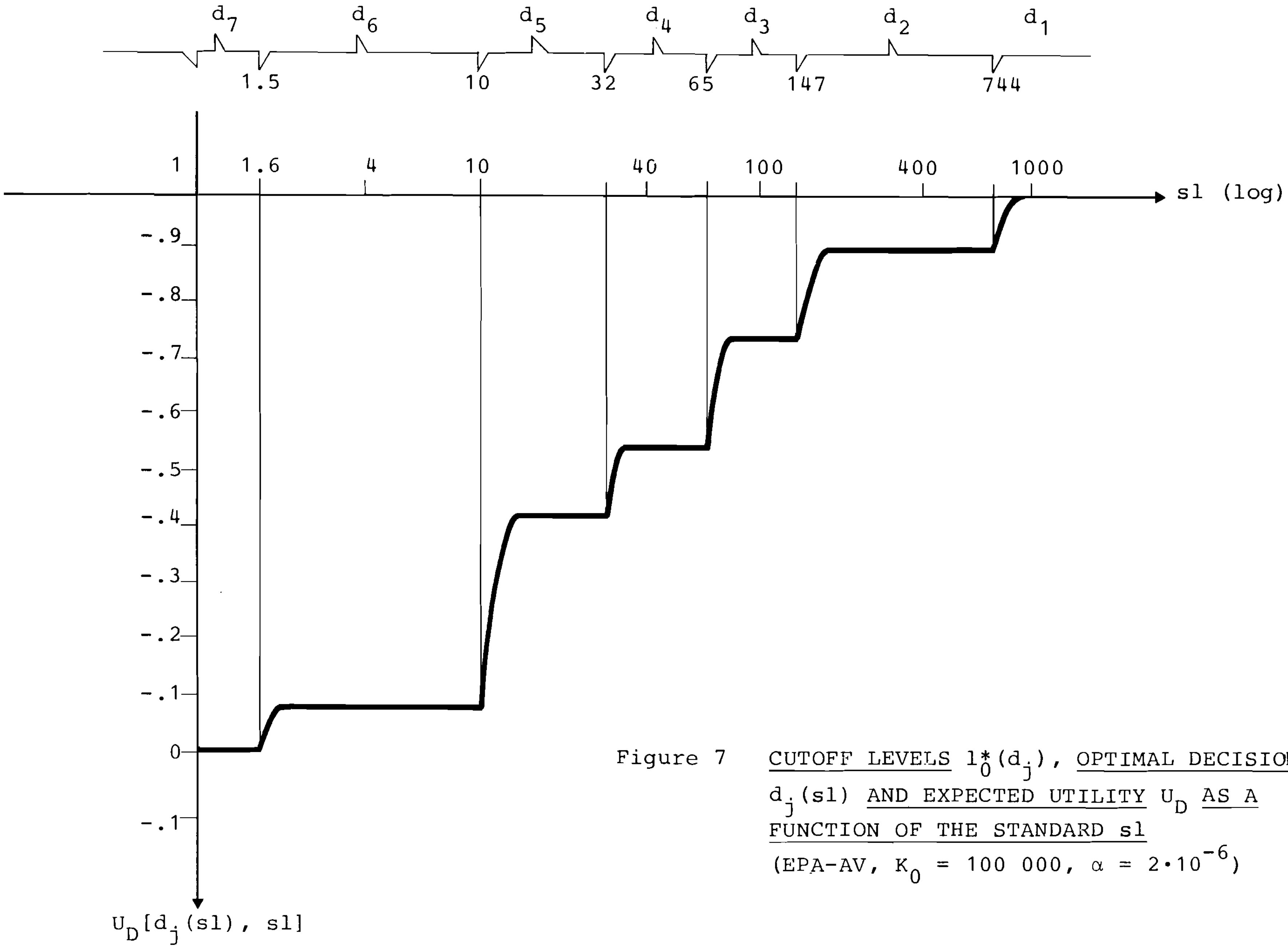

Figure 7 CUTOFF LEVELS $l_0^*(d_j)$, OPTIMAL DECISIONS $d_j(sl)$ AND EXPECTED UTILITY U_D AS A FUNCTION OF THE STANDARD sl (EPA-AV, K_0 = 100 000, $\alpha = 2 \cdot 10^{-6}$)

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:

$$v_A(l) = -l \qquad (49)$$

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 dilluted 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

$$U_A(d_j) = \int_{-\infty}^{+\infty} N(\bar{l}_j, \hat{s}_j) \cdot (\text{sign } \beta) \; e^{\beta(-l)} dl \qquad (50)$$

for $\beta \neq 0$ and as

$$U_A(d_j) = \int_{-\infty}^{+\infty} N(\bar{l}_j, \hat{s}_j) \cdot (-l) dl \qquad (51)$$

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 $\bar{I}_j$	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.

* 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

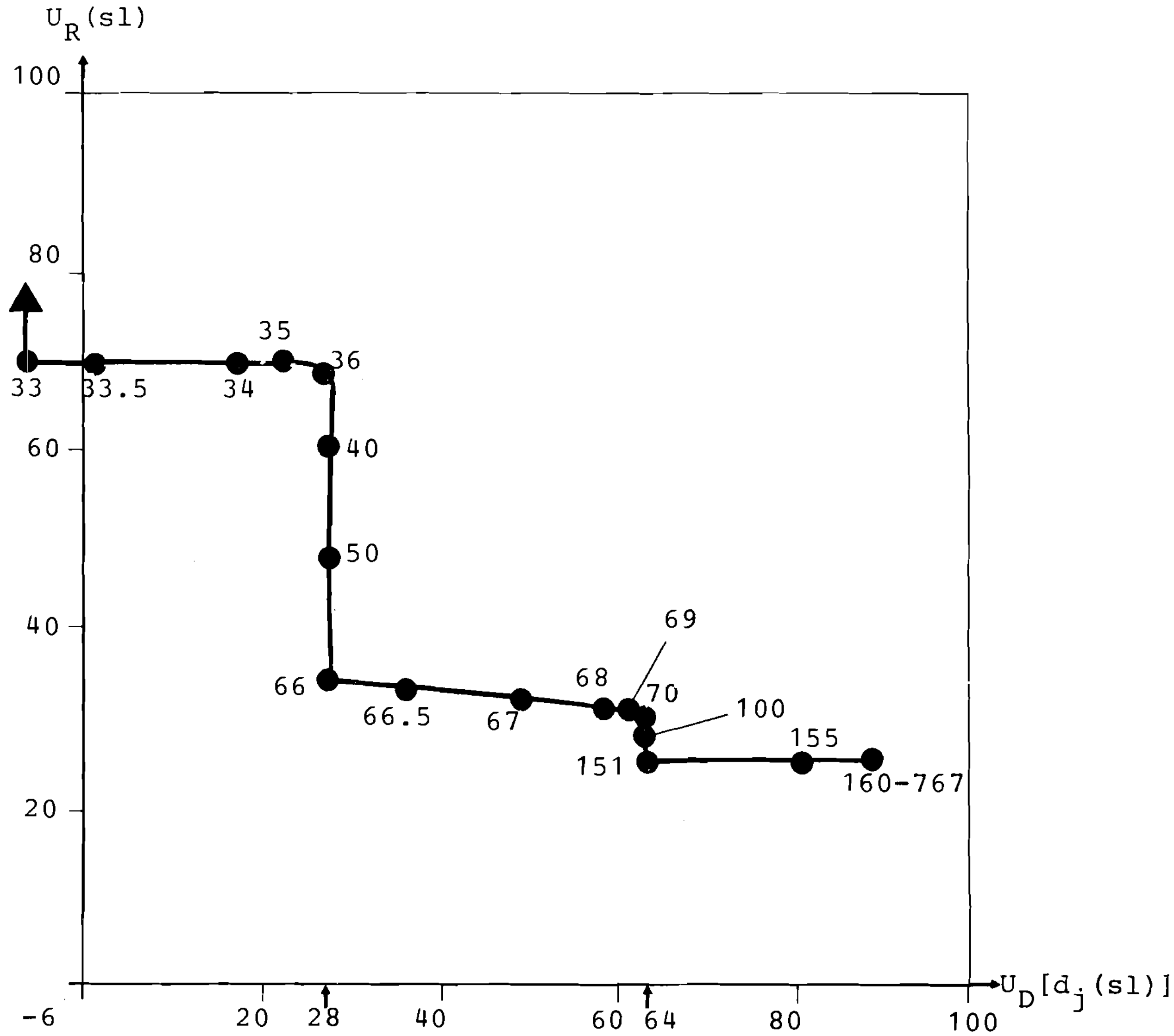

Figure 8 STANDARDS BETWEEN 33 AND 767 IN THE REGULATOR VS. DEVELOPER UTILITY PLANE
(EPA-AV, $\alpha = 0$, K_0 = 100 000, unit weights w_{Ri})

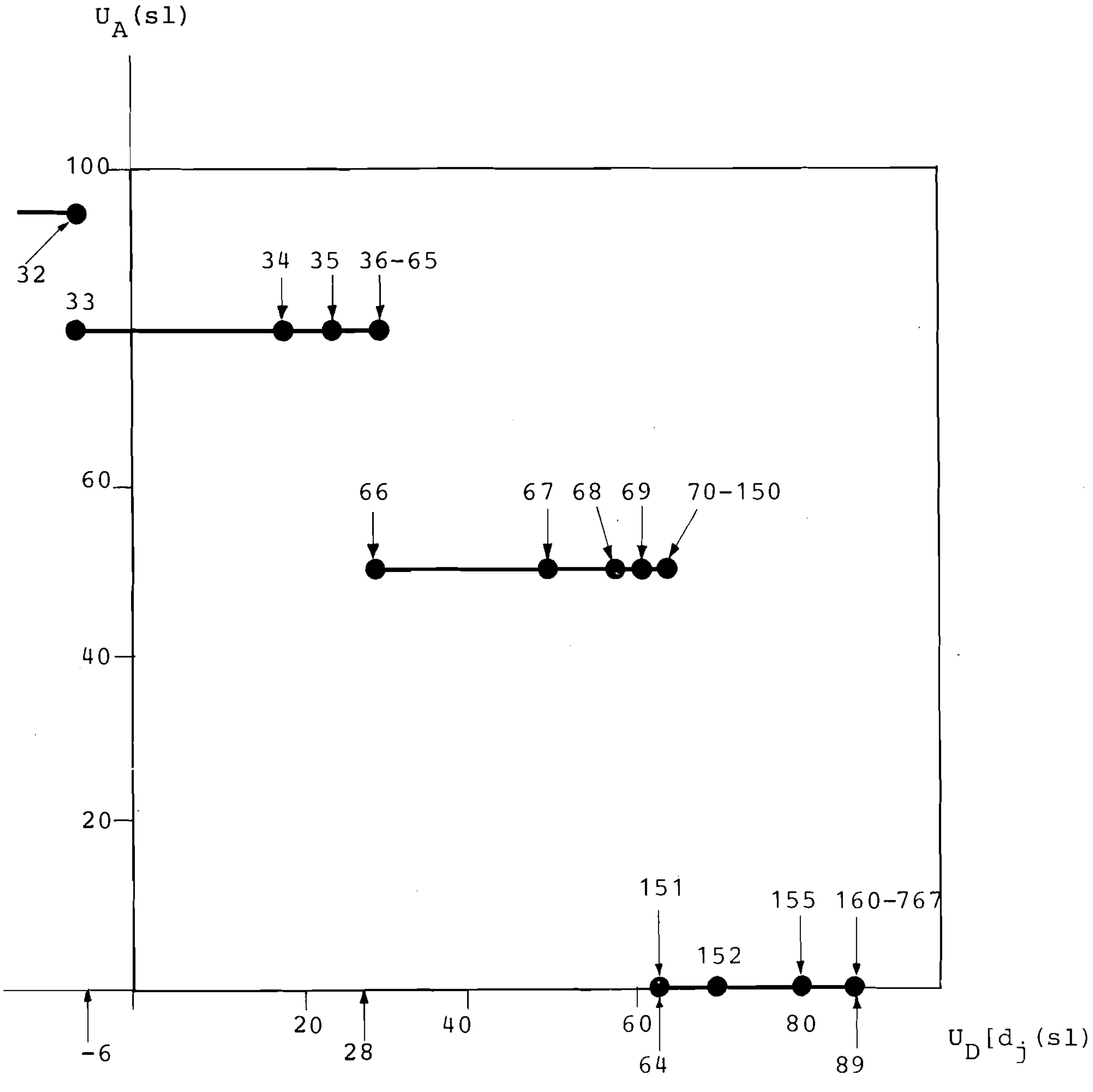

Figure 9 STANDARDS BETWEEN 33 AND 767 IN THE DEVELOPER-IMPACTEE UTILITY PLANE

(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 ($\hat{s}_j \to 0$) 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

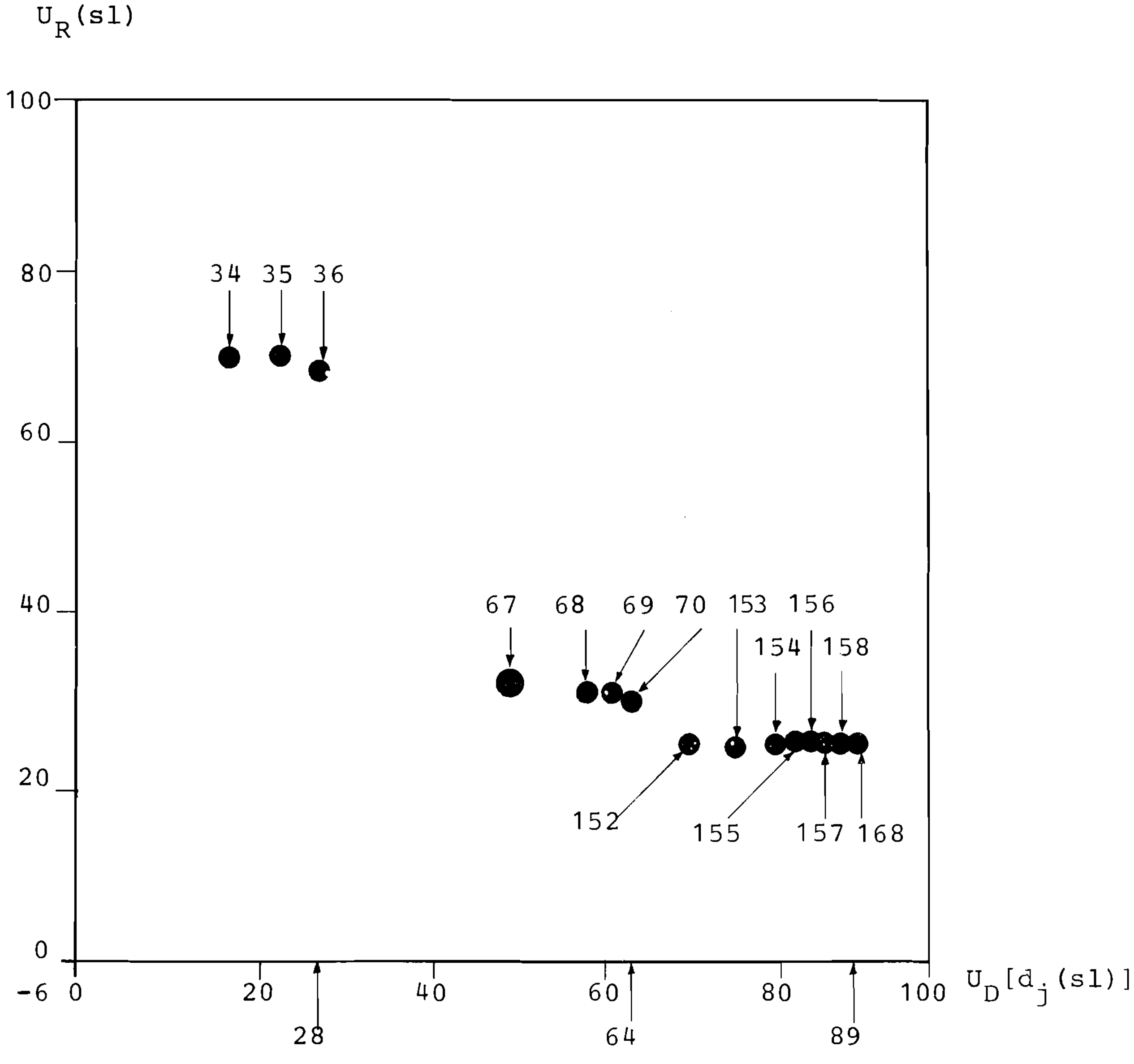

Figure 10 NON-DOMINATED STANDARDS BETWEEN 33 AND 767 IN THE REGULATOR-DEVELOPER UTILITY PLANE
(EPA-AV, K_0 = 100 000, risk neutral utilities, unit weights w_{Ri})

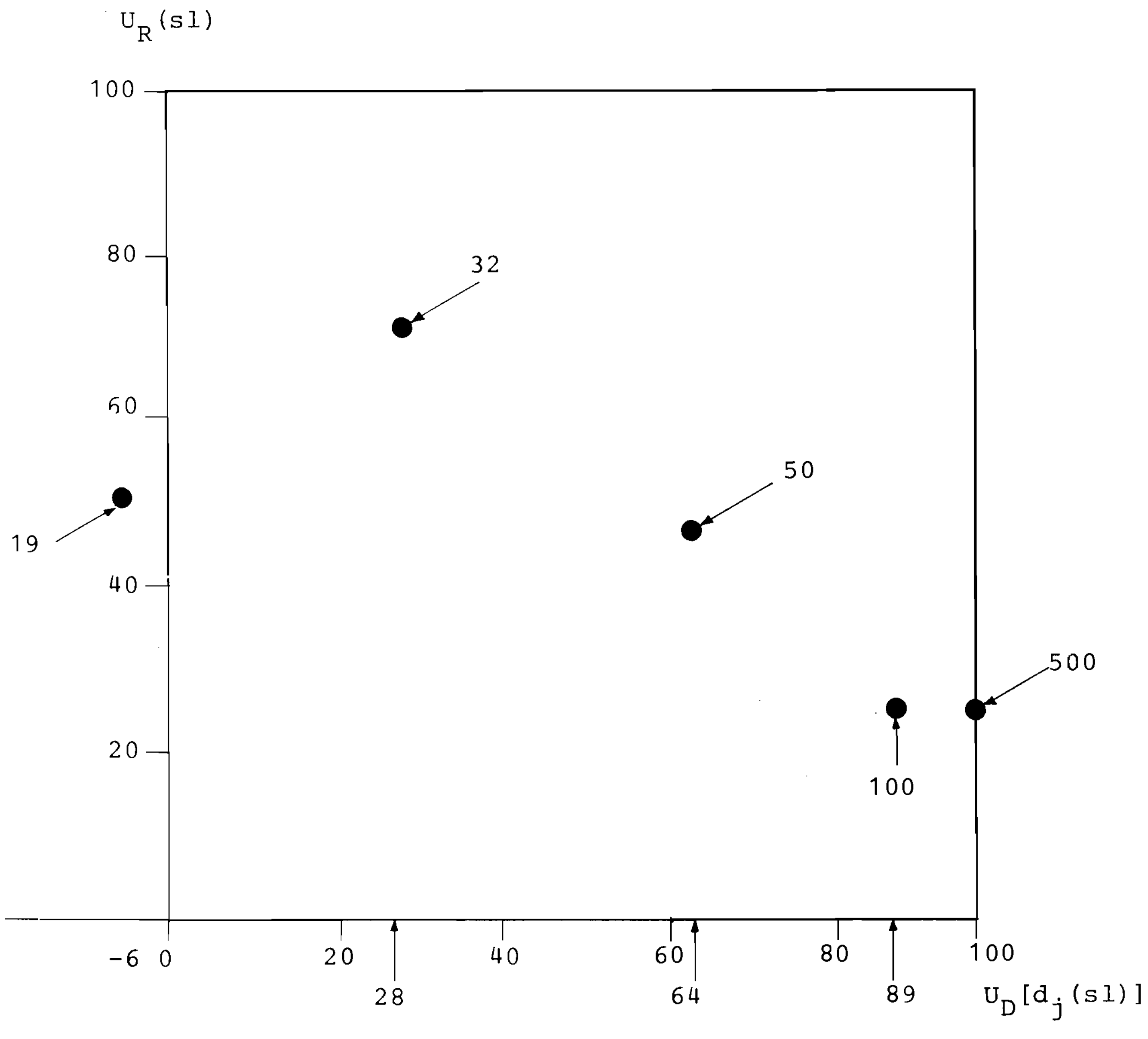

Figure 11 NON-DOMINATED STANDARDS BETWEEN 5 AND 500 PPM IN THE REGULATOR-DEVELOPER UTILITY PLANE UNDER PERFECT INFORMATION
(EPA-AV, K_0 = 100 000, risk neutral utilities, unit weights w_{Ri})

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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