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Standards and Environmental Criteria: The Practical Application of the Results of Laboratory Experiments and Field Trials to Pollution Control [and Discussion]

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Standards and environmental criteria: the practical application of the results of laboratory experiments and field trials to pollution control

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The derivation of standards and their application to the regulation of marine environmental quality are discussed, including the rôle of monitoring. The significance of experimentally determined sublethal effects in relation to the setting of environmental standards is considered. Contemporary approaches to pollution control are compared especially as they affect problems of standard setting.

Introduction

The marine environment has a capacity to receive toxic materials of all kinds. From a scientific viewpoint the problem is therefore one of defining that capacity in quantitative terms for various pollutants or combinations of pollutants and then setting appropriate controls on the introduction of polluting materials such that that capacity, or some fraction of it, is not exceeded.

The concept of environmental capacity

Environmental capacity is here used in the sense of a rate of introduction for the pollutant in question derived from an appropriate exposure standard for that pollutant in relation to the pollutant–target combination posing the greatest overall constraint on the introduction of the pollutant. In critical path terminology (I.C.R.P. 1966), as applied to environmental assessment, these are the critical pathway(s) and critical target(s) whose control will provide adequate control of the overall situation. The critical target may be man, as in the case of radioactivity (I.C.R.P. 1977; I.A.E.A. 1976), or some element of the marine biosphere, or, as in the case of oil and litter, some physical component or aesthetic quality of the environment. In setting primary exposure or protection standards it will be necessary to obviate any unacceptable degree of risk to public health or of environmental damage. However, it will seldom, if ever, be possible, or justifiable, to eliminate risk of effects altogether since cost will be an important consideration and the essential requirement will be to strike a balance between the cost of reducing effects and the value of the benefit achieved in so doing (I.C.R.P. 1973).

The derivation and application of standards

The essential steps in the derivation and application of standards can be thought of in four stages:

(1) the criterion of dose–response (see below), which relation is usually taken to be linear and without threshold because of the difficulty of demonstrating unequivocally the presence

or absence of thresholds at the low doses applicable to properly managed environmental situations;

(2) the setting of primary protection standards based on the relation derived, or assumed, in (1);

(3) derived working limits, or environmental quality standards (secondary or tertiary protection standards), based on the primary standard but interpreted, for example, in the light of data on the metabolism or behaviour of the critical target organism, or the water quality criteria or environmental quality objectives that it is desired to achieve;

(4) the derivation of discharge standards based on (3), but taking account of the characteristics of the particular receiving environment and of national policy and/or international obligations and including the considerations of cost/risk and cost/benefit.

Dose–response criteria and some of their implications for exposure standard setting

The starting point then is the development of dose–response criteria, that is, the relation between exposure and effect. Definition of an acceptable degree of exposure requires quantitative assessment of the relation between dose and effect. The effect in question may be death, or a degree of impairment of function which carries with it a lower probability of survival than would otherwise be expected. This latter category of effect would include sublethal effects in the sense of the subject matter of this meeting; that is, effects determined experimentally which are of sufficient severity to carry with them a high probability of death or reduced reproductive capacity which, in the natural struggle for survival, would have a similar significance for the exposed population as a whole as relatively prompt direct lethal effects to substantial numbers of individual members of the population.

The dose–response relation should be established if possible for different rates or levels of exposure. However, the relation is only likely to be established from effects observed at relatively high doses and dose rates. It may never be possible to develop the relation at low dose rates from direct observation of effects. It will therefore usually be necessary to assume the form of this relationship at low doses and low dose rates. The response curve in this lower dose régime may be assumed to be one of two basic types (figure 1). The curve which exhibits a threshold will in most cases for practical purposes, at low dose rates, be indistinguishable from the linear response curve, and a standard for acceptable exposure set on the basis of the linear response hypothesis will afford an added degree of protection. There will be a few cases where the threshold itself can be determined, and where this is possible then clearly there is the option of setting a standard below the threshold, that is, a no-effect level. Even where this is practicable, however, care should be taken to see that the implied costs of so doing are justified by the advantages to be derived. Otherwise the standard should be set somewhere on the dose–response curve above the threshold. How far above should be determined by cost/risk, cost/benefit considerations.

The dose–response relation may more usually be assumed to be linear and without threshold. This is a very pessimistic assumption, and attempts to establish effects at low dose rates will probably fail to demonstrate any detectable effect. It is likely, therefore, that estimates of effect based on such assumptions will in many cases be overestimates of actual effects, and will lead to the establishment of primary standards (exposure or dose limits) which may contain

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substantial margins of safety. There are two immediate consequences of this type of dose–response hypothesis, which are important when applying standards derived from it to environmental situations. One is that since any dose carries with it some finite, even if very low, risk of effect, dose or exposure should not only be below the standard selected but as far below as is practicable, bearing in mind considerations of cost. The second consideration, again since all doses carry at least an implied risk of effect, is that the collective dose to the whole of the exposed population should at least in principle be assessed in order to estimate the total detriment. These considerations will not often be of overriding importance in human populations, where limitation of risk to the individual will more usually be the aspect of major importance, but they will usually be the aspects of primary importance in the context of other organisms. This concept of collective dose has been well developed in respect of radiation to take account of its mutagenic or carcinogenic properties when human populations are irradiated (I.C.R.P. 1977). It has not as yet been applied in the context of other animal populations, though it has a potential relevance for those carcinogens, mutagens and teratogens which are not readily degraded in the environment.

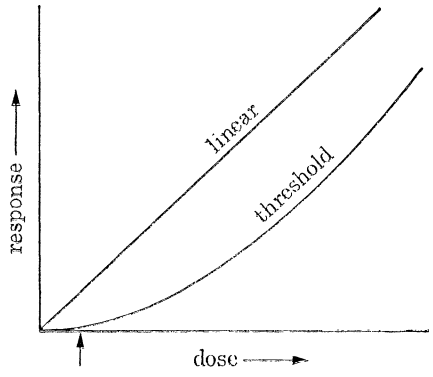

FIGURE 1. Possible dose/response relationships at low doses. The threshold dose is arrowed. Modified from Preston (1977) (I.A.E.A., Vienna).

However, the use of collective dose concepts, although in principle requiring the summation of all doses, however small, needs a great deal of care in its application and interpretation, since the summation of extremely low doses per individual delivered at very low dose rates over large population groups may give a very false picture of the risks actually being run. In natural populations, subject to large variations in natural mortality, such considerations in relation to the costs of reducing a very large number of extremely low doses will seldom if ever be justified by the benefits to be gained by so reducing exposure. There is therefore in humans, as well as in other animal populations, an implied lower limit beyond which the cost of reducing dose will be prohibitive. In the context of animal populations, cut-off might be imposed arbitrarily, as for example at levels of exposure implying say 10% of the natural variation in the occurrence of the effect being assessed, which would be undetectable and unlikely to influence the outcome of the struggle for survival in such populations.

In the absence of adequate data at the low dose rates applicable to the environment, the linear dose hypothesis should provide a basis for setting conservative primary standards and result in the derivation of sensible environmental quality standards (derived working limits) sufficient to permit reasonable control until better data are available. Such estimates would

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need to reflect the caution proper to safety and environmental protection considerations, but should not demand large and unrealistic margins of safety for every uncertainty. Furthermore, since they will often need to be established from laboratory studies or even theoretical considerations in advance of any direct evidence of their validity – indeed, as already noted, it may in many cases never be possible to establish acceptable levels on the basis of observed effects under natural conditions – they should be treated with some flexibility and not regarded as rigid lines of demarcation below which there is no effect and above which effects automatically become manifest.

The crucial question therefore is what level of effect, implied by the choice of a particular point on the dose–response curve, is acceptable? It has already been suggested that zero effect is not a practicable objective and we know that in the majority of cases we are dealing with experimentally derived data whose relevance to real environmental considerations is questionable. In human health terms, as for example with radiation, the choice has been exercised on the basis of prevention of threshold effects in the individual, i.e. those whose severity is a function of dose, and reduction of the risk of stochastic effects, i.e. those whose frequency of occurrence is a function of dose, e.g. carcinogenesis, to an acceptable rate of occurrence in the exposed population as a whole. With marine organisms, it is unlikely that we would be unduly concerned with the protection of a few individuals but rather with the health of the population as a whole, i.e. stochastic effects. We might thus wish in this case, as already suggested, to set the standard at a level where the probability of occurrence of effects was undetectable in the environment against the background of spatial and temporal variability or the effect of some other imposed stress such as fishing mortality. D. H. Cushing (this symposium) has emphasized that care needs to be exercised in this latter context where the populations are fished beyond their maximum sustainable yield (m.s.y.). It might even be suggested that optimum use of the environment for waste disposal, where the probable effect is on the wellbeing of an exploitable fish stock, demands management of the fishing effort at levels approximating the m.s.y.

We have heard during the course of this symposium something of the difficulties of investigating toxicity on an experimental basis in relation to individual organisms, populations and ecosystems and of the problems of extrapolation to the environment. Equally, the monitoring of environmental situations to determine the presence or absence of an effect attributable to pollution presents major difficulties against the background of normal variation. Furthermore, it is probable that at levels of pollutant concentration likely to be deemed acceptable in the environment we will seldom be able to demonstrate effect in unequivocal terms except in the immediate vicinity of major disposal operations. It seems much more realistic therefore, for the time being, to approach the problem of monitoring, and related toxicity and standard setting problems, on the basis of chemical residue monitoring in the critical media, organism, water or sediment as appropriate, exercising value judgements on the data obtained by comparison with environmental quality standards based on a primary protection standard derived from a dose–response relation which in part is built up from toxicity testing at relatively high exposure rates. This will mean, among other things, that toxicity testing will have to be concerned with the establishment of body-burden data and the metabolism of pollutants in relation to the endpoints selected for the test itself. The identification of critical organisms and organs and their associated pollutant burdens will be necessary in order to ensure that the monitoring effort is properly directed and the data from it interpretable.

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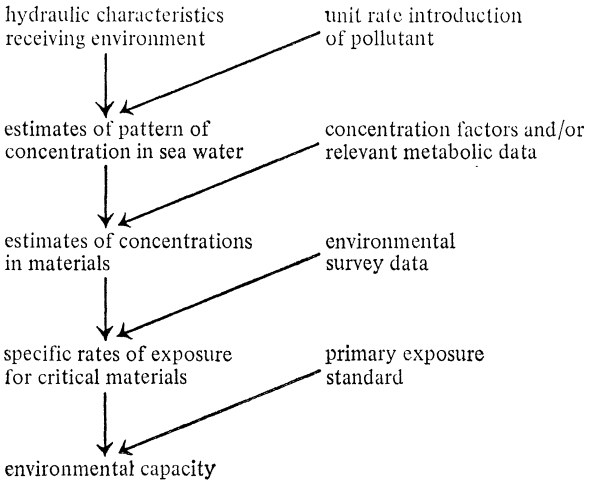
Thus a wealth of experimental work remains to be done, even in relation to the commoner pollutants, if we are to establish responses at a series of exposure levels and associate them with pollutant concentrations in the relevant critical organ in a representative selection of marine organisms of demonstrated sensitivity and ecological relevance.

The derivation of environmental capacity

The standards, once developed, can be used to control specific environmental situations by deriving limits to the environment's capacity which are based upon them. Environmental capacity is here taken to be the rate of introduction of a pollutant which at equilibrium will result in a rate of exposure of the critical target(s) per unit time equal to that defined by the primary standard.

Table 1. Principal stages in pre-disposal assessment of environmental capacity



The preferred method for evaluating environmental capacity will be based on critical path techniques and will seek to estimate the equilibrium levels in critical materials resulting from unit rates of introduction of the pollutant in question (table 1). The assessment of capacity in this way will require a careful environmental study to obtain some idea of the pollutant's pattern of behaviour and general distribution after release, especially in relation to those uses, or intended uses, of the receiving environment that will set the basis for environmental quality objectives. These studies are a prerequisite to the establishment of acceptable rates of introduction and they will also indicate whether there is a need for environmental monitoring and if so will permit its planning and execution in the most effective way.

The scale of investigations required will vary enormously from one situation to another, according to the amount and toxic nature of the material requiring disposal, to the physical and dispersal characteristics of the environment to which it is to be introduced, and to the uses made of that environment; it is therefore essential to conduct investigations on a case-by-case basis. Where the major effect is to cause a change in the physical conditions of the environment, or an accumulation of undesirable substances in the food chain leading to man, the

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selection of critical materials presents no serious problems, but where the major effect may be on the stability of populations or whole ecosystems, substantial pre-discharge investigations may be required to establish the critical routes and materials.

Such environmental surveys will also need to be concerned with a detailed assessment of the uses made of the environment which might lead to human exposure either directly or via the food chains, and the principal pathways will need to be quantified. In cases where the pollutant is expected to exert its most significant deleterious effect on some other component of the environment, detailed consideration will have to be given to the populations or to the ecosystems in question, and this may require extensive investigations.

During the conduct of these environmental surveys those segments of the exposed populations likely to receive the greatest exposure will be identified; these will usually be small groups. However, in a few instances the populations exposed may be large and it will then be necessary to establish, for a representative sample of the population, those behavioural factors that will influence the degree of exposure, for example human consumption habits, migratory patterns, and life-stages and life-span of other organisms. It will normally then be possible to isolate a group whose behaviour makes them liable to receive the greatest exposure. Rates of introduction of the pollutants may then be set so that the average exposure of this group does not exceed the primary standard. In the case of animal or plant populations, the death of or damage to a few individuals will not normally be regarded as of paramount importance, and the need will be rather to protect the population of an area and to maintain, so far as it is practicable, the stability of the ecosystem, the productivity of exploitable resources or the amenity characteristics of the environment.

It should be the practice of the regulatory authorities to set rates of introduction well below those compatible with the limiting capacity, and, during this early period of discharges, the opportunity should be taken where appropriate to check the accuracy of the original assessment through the monitoring of effluent composition and the environment.

Monitoring

Once acceptable rates of introduction have been arrived at, the control framework within which continuing discharges are made will necessitate some degree of monitoring (Preston & Wood 1971). Monitoring is here used to mean the measurement of a pollutant or its effects, for reasons related to the assessment or control of exposure to that pollutant of either man or specified components of the natural environment. Two categories of monitoring, the rate of input and the levels or effects in the environment, may be distinguished; the former in any case should be a regulatory requirement.

Monitoring of the levels of pollutants in the environment might in some cases start with a baseline survey to establish existing levels, possibly to be followed by some form of regular monitoring. A baseline survey will be of particular importance where previous introductions of the pollutant have not been preceded by an environmental assessment or accompanied by any necessary controls, e.g. polychlorinated biphenyls and DDT. The objectives of environmental monitoring should be related to the assessment of the actual or potential exposure of the critical target(s) resulting from the introduction of the pollutant, or, in some cases, estimation of the probable upper limit of such exposure. It may be possible on occasion to look for the actual effects of pollution: so-called 'biological' or 'ecological' monitoring. The proper

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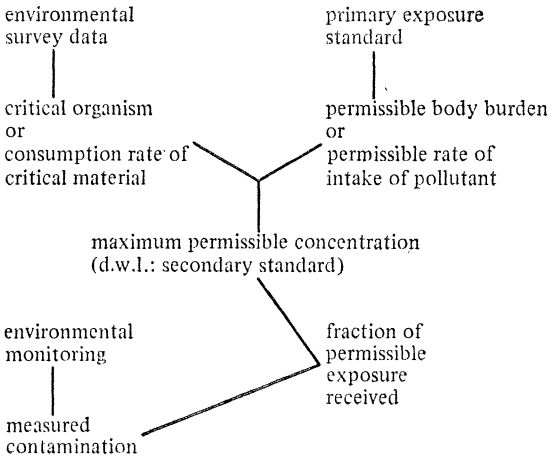
implementation of environmental monitoring programmes envisages a much better quantitative understanding of the toxicity, effects and associated body burdens of many pollutants than we have at present.

The pursuit of monitoring operations for their own sake, without the ability to interpret the significance of the data obtained, i.e. by comparison with derived environmental quality standards, will seldom be a profitable use of resources except possibly to establish spatial and temporal trends in which case the frequency of survey can be substantially reduced compared with that of a routine regulatory monitoring programme. In any case, all monitoring programmes should be subject to periodic review, particularly from the point of view of the environmental implications of releases.

Thus all surveillance operations should be considered as two-stage processes. The first phase, the monitoring of discharges, should be mandatory. The second, environmental monitoring, needs only to be established where it is justified in relation to the nature and scale of the discharge. In general terms, and for the majority of pollutants, there has in fact been more effort on environmental monitoring than on input monitoring and this has led to the situation where observed levels in the environment cannot be sensibly related to rates of input. This is a regrettable loss of opportunity to establish a basis for sensible regulation of input.

TABLE 2. DERIVATION AND USE OF D.W.LS FOR INTERNAL EXPOSURE



DERIVED WORKING LIMITS OR ENVIRONMENTAL QUALITY STANDARDS: SECONDARY AND TERTIARY STANDARDS IN RELATION TO THE CONDUCT OF MONITORING OPERATIONS AND ENVIRONMENTAL QUALITY CONTROL

Direct measurement of the exposure of targets to pollutants is seldom possible as part of a monitoring operation, and resort has to be made to measuring exposure indirectly, by using measurements of contamination in conjunction with environmental survey data such as food consumption rate or occupancy factor. The method employed is to calculate derived working limits (d.w.ls), sometimes referred to as environmental quality standards. They are based on the primary standards but do not have either their validity or wide applicability, but are based on the case study for the particular segment of the environment for which the assessment was carried out. The methodology applicable to the derivation of these secondary standards

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for internal and external exposure to pollutants is given in tables 2 and 3. The tables also demonstrate how they are used in a monitoring operation, i.e. to derive estimates of exposure to the critical individuals or population groups. The second important use of monitoring data is the quantitative post-operational reassessment of the environmental capacity (table 4), where both details of effluent composition and environmental monitoring data are required.

TABLE 3. DERIVATION AND USE OF D.W.LS FOR EXTERNAL EXPOSURE

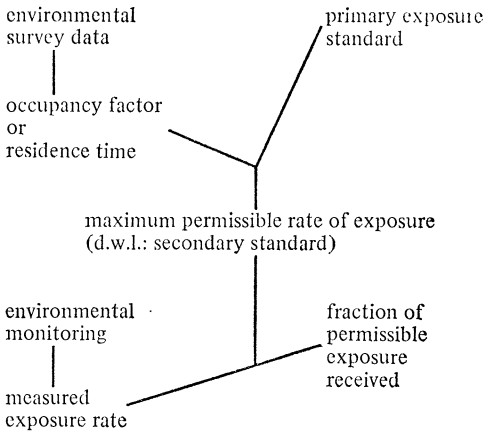
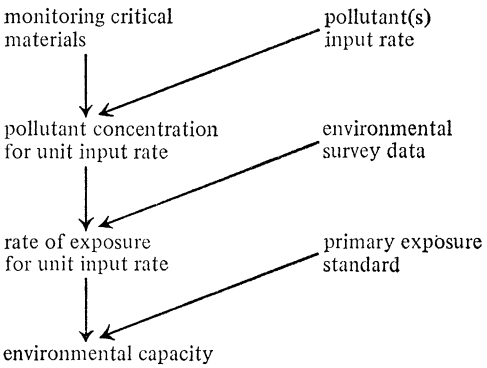


TABLE 4. PRINCIPAL STAGES OF POST-DISPOSAL REASSESSMENT OF ENVIRONMENTAL CAPACITY



There are of course other methods of applying basic, or primary, standards to environmental quality control. Though they need not be, they are often applied in a less environmentally realistic manner than has been suggested for d.w.ls. There is, for example, an increasing tendency to apply uniform quality standards to a whole medium, e.g. sea water, in order to achieve a given environmental quality objective. There are also attempts to impose a policy of uniform emission standards which almost totally ignores the disparate nature of different receiving environments.

It is crucially important to remember that basic, or primary, standards are based on criteria developed in relation to particular targets, and either are derived on the basis of a linear response type hypothesis as a basic protection level or, where a threshold in the dose–response relation can be demonstrated, are sometimes set immediately below that threshold as a no-effect level. In the case of a basic protection level, i.e. where a target is not exposed to unacceptable risk, an element of cost/risk and cost/benefit is involved in the judgement as to what

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constitutes an acceptable risk. This alone implies some flexibility of approach and is therefore itself contrary to the inflexible use of arbitrarily derived exposure or concentration standards, which are based on primary standards but take no account of individual environmental circumstances. This insistence on uniformity of quality standards rather than on uniformity of application of primary standards assumes that application may be made, without further modification or interpretation, to a wide range of environments. It is this latter inflexibility of approach which tends to be seen in attempts to regulate environmental levels of many pollutants and which appears to ignore the experience gained, as for example with radioactivity. In the majority of cases, each environmental situation will be different, and the values of the variables which will determine the ultimate exposure of a target will differ. A wide range of actual values for environmental quality standards will result, each related to a particular environment, but each will therefore suffice to achieve the overall objective of protecting the target to the same extent, i.e. the extent determined by the original selection of the primary standard. As has been shown, these variables, whose interactions determine the ultimate exposure of the target, include physical, chemical and biological factors, and no *a priori* assumptions about their individual values or interactions will satisfy the conditions of every receiving environment.

Again it must be remembered that basic or primary standards are derived against accepted dose–response criteria to ensure so far as is reasonably achievable that there is no undue exposure of a target. Standards based on these primary standards (i.e. secondary, tertiary, etc., standards) are set so as to ensure that in particular sets of circumstances, where compliance with the primary standard cannot be directly shown, there is reasonable assurance that it cannot be exceeded. Provided it is not exceeded, the derived standard based on the primary standard should not seek to impose unnecessary further safety factors. This will surely be so, however, if uniform quality standards are insisted on, since in some environments the other factors determining the exposure of the target will be such as to lead to very much smaller exposures than they will be in the more restrictive environments. Unnecessary degrees of either over or under protection will thus be imposed at considerable cost, depending on whether the quality standard is set for the more or the less restrictive situation. It is for these reasons that each environmental situation requires separate evaluation to enable quality standards for a particular environment, appropriate to the desired quality objective(s), to be derived. What is required, therefore, and what regulatory authorities should be directing their attention to, is the establishment and adoption of a common approach to environmental assessments which will permit the uniform application of basic or primary standards; this is certainly not the same as uniform values for quality standards, let alone uniform emission standards.

It is perhaps true that in a legislative or administrative sense the provision of uniform quality standards would greatly simplify the regulatory process, and facilitate aspects of harmonization. To some extent, for some pollutants in some media, it is possible, and perhaps sufficiently worthwhile, to meet this concept of uniform quality standards and to provide broad figures (ranges) of acceptable validity, as for respirable air or drinking water, but this is so solely by virtue of the fact that, for the requisite degree of accuracy, people may be assumed to breathe the same quantity of air or drink the same quantity of water. The essential points here are that the standards are set for the consumption of the medium itself by the exposed targets, and that every target is deemed to suffer the same degree of exposure. This will not always be true even for these media, and the moment these implicit constraints are removed, as for example when

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these concepts are extended to the exposure of a target via some secondary or even tertiary route, e.g. in food consumption, the increased variability introduced to the situation by virtue of the extra links in the exposure chain makes nonsense of the application of uniform quality standards to the primary (receiving) medium.

This emphasizes a very important regulatory principle in environmental situations, that is that the monitoring operation and the standard to which it refers should relate as closely as possible to the point at which damage to the target, as a consequence of exposure, is likely to occur.

TABLE 5. E.E.C. ENVIRONMENTAL PROGRAMME TERMINOLOGY

(Environmental quality standards are regarded as one method of helping to construct and reach environmental quality objectives.)

criteria
(developed from a dose–response model)

↓

primary standard
(body/organ burden or exposure rate at either a basic protection level
or
a no-effect level)

↓

secondary/tertiary standards
(environmental quality standard; if the standard relates to the medium in terms of direct exposure to the target it is a secondary standard; it is otherwise a tertiary standard)

CRITERIA, PRIMARY (BASIC) STANDARDS, QUALITY STANDARDS (D.W.LS) AND QUALITY OBJECTIVES

Some of these concepts and adaptations of them have been adopted within the aims and objectives of the Environment Programme of the European Communities (Commission of the European Communities 1974), which defines the concept of criteria (dose–response relation) and refers to standards in the context of quality objectives. It recognizes that a standard based on the developed criteria may be either (*a*) a basic protection level, where the target is not exposed to unacceptable risk; or (*b*) a no-effect level, no identifiable effect on the target. Either of these might be regarded as primary standards in appropriate circumstances (table 5). It is further recognized that 'proper account must be taken of the specific characteristics of the regions in question' when using these primary standards or standards based on them to achieve environmental quality objectives. The overall aim in the use of these concepts is to protect human health and safeguard the natural environment. Thus far there is agreement in purpose and initial methodology between the concepts of the Community Programme and the application of critical path analysis as already described in this paper. However, at the point where the primary standards are interpreted in environmental terms, differences appear. In the use of critical path analysis it is essential in applying standards to take due account of environmental variables, a procedure which the E.E.C. programme, in its implementation so far, appears to ignore, except perhaps in broad regional terms. These variables will determine the degree of exposure experienced by the target, and the well worked example is that of the application to exposure of humans to ionizing radiation, where the primary standards are the acceptable degrees of human radiation exposure recommended by I.C.R.P. and widely adopted

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within national regulatory bodies and within the E.E.C. itself in the Euratom Directives (table 6).

To apply these primary standards for human radiation exposure to the radiation from internally deposited radionuclides a metabolic model of a 'standard man' has been arrived at, and human organ or body burdens of individual radionuclides have been established from this model such that these radiation exposures will not be exceeded. These burdens are secondary standards, and in so far as individual metabolism will vary from that of the 'standard man', a range of exposures will be experienced in any actual population of individuals exposed to such levels.

TABLE 6. IONIZING RADIATION: CRITICAL PATHWAY TERMINOLOGY

criteria
(dose–response or exposure–effect relations)
↓
primary standard
(exposure rate)
↓
secondary standard
(body/organ burden)
↓
tertiary standard
(daily intake to establish and maintain body burden)
↓
quaternary standard
(concentration in medium or product consumed to match daily intake limits)
↓
quinary standard
(concentration in water or air to match concentration in product, e.g. fish)

These body or organ burdens are in turn related to rates of intake sufficient to establish and maintain the burden. These rates of intake are tertiary standards. They may in turn be translated into concentrations in air or water based on the simplifying assumption that humans have a uniform intake of air and water: these concentrations in air or water are quaternary standards and they will clearly be another source of variation within any real exposed population. In the case of intake from food, no such simplifying assumption can be made about intake, and individual food consumption rates for particular foods in particular circumstances have to be established in order to set up quaternary standards for various foods, many of which will be so closely related to local circumstances as to be site specific.

In the context of, for example, marine foods, the establishment of standards for concentrations in sea water which will lead to the acceptable concentration in foods such as fish and shellfish, that is, the establishment of quinary standards, requires a host of other variables to be taken into account, including the chemical and physical states of radionuclides which will determine their entry into plants and animals used as foodstuffs. Thus, the establishment of such standards introduces another set of variables leading to an even wider variation of exposure in a real

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exposed population of consumers. It thus becomes even more important to tailor the application of primary standards to individual environmental circumstances.

The common methodology for application of standards has been largely met, for radiation exposure from environmental contamination by radioactive materials, by the application of critical path techniques to the evaluation of individual environmental situations. I.C.R.P. (1966) has recommended the identification and establishment of a critical group of exposed individuals the control of whose radiation exposure will set acceptable limits to environmental contamination. Clearly, the value of specific variables determining the degree of exposure of such groups will vary from critical group to critical group and specific case studies will nearly always be required for each environment. It is thus evident that the establishment of widely applicable environmental quality standards could not be met in this type of situation except as already noted in the case of drinking water or air. Furthermore, it would defeat the basic concepts of radiation protection that exposure should be held to a practicable minimum taking due account of cost/risk and cost/benefit considerations, since individual consideration of risks and benefits is required to implement such a philosophy, that is, specific case studies at each situation. Similar considerations will apply to metal contamination and many other pollutants, not only in the context of human health but also in the context of environmental damage. What is required therefore is the establishment of a sensible rationale or framework that will permit a meaningful and reasonably uniform application of primary standards to specific situations or cases. Such a framework seen against the background of environmental quality objectives will have to take due account of many environmental variables as well as those associated with human habits. The application of critical path techniques probably affords the best opportunity to provide such a framework (I.M.C.O. *et al.* 1976).

The absence of widely acceptable standards derived on an agreed and rigourous scientific basis has undoubtedly made the design and particularly the implementation of control procedures very difficult, and many arbitrary decisions have been made, often in response to public or political pressure, that might have been avoided had there been a firm base of standards to fall back on.

The appearance of such concepts as prohibition of discharge, uniform emission standards and special area concepts are in their own way symptoms of the same problem, lack of soundly based and widely accepted standards. There are no toxic materials intrinsically unsuitable for introduction to the environment. There is a finite quantity of any material that can be safely introduced: the problem is to define this quantity, and this is an almost insuperable task without identification of critical targets and definition of standards to control their exposure. With standards, the concept of special areas falls, because special features will be taken into account in setting rates of introduction based on the standards. All environments will be treated in the same way by a common method of application of agreed standards. Rates of introduction so derived will meet the differing constraints of different areas, including the so-called special areas.

Conclusions

Regulation of the introduction of potential pollutants to the environment clearly demands the identification of critical targets, the definition of exposure standards to protect these targets, and a rational application of these standards to real environmental situations.

The way ahead is reasonably clear: develop standards, if necessary on a purely empirical

basis, with the use of a hypothesis such as that of linear dose–response, unless threshold phenomena can be clearly shown to exist; apply the standards to individual situations based on the application of critical path techniques; define acceptable rates of introduction and associated environmental concentrations; monitor introduction and, where appropriate, the environment, by using d.w.ls (environmental quality standards) as the field criteria demonstrating compliance with standards.

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Discussion

R. J. Pentreath (*Fisheries Radiobiological Laboratory, Hamilton Dock, Lowestoft NR*32 1*DA, U.K.*). With regard to toxic concentrations of pollutants in water. Mr Preston has drawn attention to the need to relate these to body burdens. The majority of such toxic concentration data are derived from $l.c._{50}$ experiments and it is of interest to view these in relation to possible causes of death. A pollutant such as a heavy metal could kill in a variety of ways: it could be assimilated and cause death by interfering with various biochemical pathways; it could be inhaled and cause death by choking; or it could even kill by physical impact. The majority of aquatic $l.c._{50}$ tests probably relate to the second analogy for metals, with very little assimilation, whereas the majority of sublethal effects – the topic of this meeting – are probably the result of the first. The environmental data which approximate to body burden data, however, have been largely derived from programmes aimed at monitoring the edible fractions of marine organisms. The two sets of data, $l.c._{50}$ concentration in water and edible fraction concentration, are not clearly related one to another. It would appear that what is required is first a knowledge of which organs respond the most to an increased intake of pollutant from enhanced concentrations in the water, and secondly how enhanced concentrations in organs relate to demonstrable effects – lethal or sublethal – in laboratory studies. It may then be possible to monitor specific organ concentrations of pollutants in contaminated areas and to interpret these with regard to their possible environmental effect. This is by no means an ideal approach but would at least be based on a measurement which is capable of some degree of verification, i.e. chemical analysis which, as Mr Preston has said, may well be the only practical method.

K. W. Wilson (*North-West Water Authority* (*Rivers Division*) *Buttermarket St., Warrington, Cheshire, U.K.*). Mr Preston suggests that monitoring of effects should be restricted to the

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monitoring of residues in critical species or critical tissues previously identified by laboratory experiment. This can be seen to apply to conservative substances, but would Mr Preston comment on how the effects of non-conservative substances be monitored, and elaborate on how substances would be selected for, or seen to be in need of, monitoring?

A. Preston. The identification of chemical residues need not be restricted to the form in which the original pollutant target challenge took place. Indeed, the form in which the pollutant exerts its significant toxic effect may well be different due to a metabolic or some other change effected in the target organism itself. Experimental approaches will therefore need to identify not only the critical organ but the form in which the pollutant is most easily identified and quantified in relation to the effect regarded as critical, e.g. metabolites of DDT and egg shell thinning in raptorial birds.

The second question is much more difficult to reply to: hopefully the current approach to development of screening procedures for materials likely to have an environmental use, or use in which significant entry to the natural environment is possible, will identify those materials with marked potential for adverse effects on environmental quality. These would obviously be prime candidates, as would any potentially noxious materials whose production and use statistics indicated the likelihood of significant entry to the environment. Considerable investigative work might of course be necessary before a decision on monitoring needs could be taken.

R. J. Morris (*C.U.E.P., Department of the Environment, London, U.K.*). I agree with Mr Preston that environmental quality objectives on a case by case basis should be our objective. The problem arises with those pollutants which have a long environmental half-life. Often the production of a scientifically valid environmental quality objective, which can then be used to give a meaningful environmental quality standard, is not always possible. The Department of the Environment pollution paper no. 11, *Environmental standards*, clearly recognizes that the environmental hazards of potentially harmful pollutants can only be evaluated when sufficient data are available. Until an adequate data base has been established for each pollutant of concern it is difficult to know which is the critical organism on which the environmental quality objective for that particular pollutant should be based. Further, without adequate knowledge of the fate and behaviour of that pollutant in the aquatic system it is difficult to predict the form in which it will exist in the environment and hence the type of effects it will have on ecosystems.

A. Preston. As Dr Morris states, the problems are more difficult with very long-lived materials if only because the variables determining their interactions with environmental media are more likely to change the longer the time available for change. There are in practice few materials in this category, apart from metals and long-lived radionuclides such as ^{129}I and ^{239}Pu. The problems are not insuperable, and, as experience with radionuclides of half-lives up to 30 years has shown, satisfactory empirical approaches can be developed, some of which will necessarily have to incorporate safety factors adequate to allow for a degree of uncertainty in the assumption employed.

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