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EFFECT-RELATED EFFLUENT CRITERIA FOR POLLUTANTS

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Abstract - Estimation of hazard to aquatic life and the environment due to use and disposal of metals and chemical substances is hampered presently by the lack of a systematic process for generating and evaluating effect-related criteria for pollutants. The primary thrust of this manuscript is to provide a strategy for determining environmental risk (i.e., the probability of harm) from actual or predicted concentrations of a chemical in the aquatic environment.

INTRODUCTION

My assignment for this symposium is to cover pollutants from metal, metal working, and chemical industries. The strategies discussed, therefore, will focus primarily on point source discharges into aquatic ecosystems. Since the majority of factories in industrialized countries is located primarily on rivers or estuaries, I will further emphasize moving water systems, although not to the exclusion of lakes and oceans. The primary thrust of this discussion is to provide a strategy for determining environmental risk (i.e., the probability of harm) from actual or predicted concentrations of a chemical in the aquatic environment. "Safe" concentrations are those either producing no adverse biological effects or those for which the risk is acceptable because of benefits associated with the use of the chemical. Thus, hazard assessment or risk analysis requires both scientific judgment based on evidence and a value judgment of society and/or its representatives. A hazard assessment designed to estimate risk to an aquatic ecosystem requires evidence to make a scientific judgment on: (a) toxicity - the inherent property of the chemical that will produce harmful effects to an organism (or community) after *exposure of a particular duration at a specific concentration*, and (b) environmental concentration - those actual or predicted concentrations resulting from all point and nonpoint sources as modified by the biological, chemical, and physical processes acting on the chemical or its byproducts in the environment.

Civilization is presently in a transitional stage somewhat comparable to the agricultural revolution. The latter was caused by the simple fact that hunting and gathering of roots, berries, and so on did not produce food in sufficient quality and quantity at all times to meet society's needs. In short, the unmanaged environment was not providing nourishment that met society's expectations. The ever increasing reports of ecological damage indicate that in many localities, waste loadings have exceeded the capacity of natural systems to assimilate and transform these wastes without themselves coming to harm. A comprehensive mangement program is needed to optimize the benefits of both an industrial society and healthy natural ecosystems. Since water is used for multiple purposes (domestic, industrial, agricultural, and recreational), impairment of quality is likely to cause severe economic dislocations as well as being aesthetically displeasing and environmentally harmful. These facts have been so well established that they need not be repeated here. The primary concern is how to use the assimilative capacity of natural systems for industrial wastewater discharges without degrading the systems.

The basic management problem can be stated quite simply. Industrial waste discharges are not constant but vary enormously in both quality and quantity due to various production conditions and so on. Similarly, the assimilative capacity of an ecosystem is not constant but varies seasonally according to flow conditions, temperature, and a variety of other factors, including water quality. For example, the acute toxicity of zinc to fish is much greater in soft water than in hard water. In short, environmental quality does mediate the toxic response. Unfortunately, variations in assimilative capacity and industrial waste discharge quality and quantity are not naturally synchronized because the industrial system is driven by a marketplace economy while ecosystems observe natural cycles. Fixed arbitrary concentrations of chemicals applied by regulatory agencies over a wide area take into consideration neither regional nor seasonable variability in assimilative capacity. As a consequence, applying a single arbitrary number over a vast region, such as the United States or Europe, will overprotect at certain times in certain regions and underprotect at others. The dilemma of fitting waste discharges to assimilative capacity of natural systems can be overcome by: (a) acquiring

scientific information necessary to make a hazard evaluation in a systematic way, and (b) providing a feedback loop from the environment to determine the accuracy and reliability of the estimate made.

BENEFITS TO INDUSTRY

Other than conforming to the law, why should an industry determine biological effects - an expense only recently carried by most industries and certainly not characteristic of industries since the beginning of the industrial revolution. A primary reason is that water is no longer a "free good" (one which is unlimited in supply so that the demand never even approaches the supply). The freedom from the costs characteristic of the past, when it was more rational to consider water as a free good, is no longer tenable. In addition, the requirement for biological monitoring appears in the amendments to the Environmental Protection Act of 1972. The Toxic Substances Control Act signed into law by President Ford as one of his last responsibilities requires evidence of hazard to human health and environment. Biological assessment is clearly one of the types of evidence that will indicate on an ongoing basis that desirable water quality conditions are being maintained. It is also worth noting that even the most expensive computer interfaced biological monitoring systems cost only a fraction of the cost of a modern waste treatment system. Industries which have comprehensive biological effects studies should get benefits from the regulatory agencies not available to those industries that lack such systems. These benefits might take the form of being allowed to exceed national and state regulatory standards when it can be demonstrated with substantial evidence that no biological degradation occurs as a consequence. Precedent for this is in the regulations published by the Environmental Protection Agency (1) which set forth the various types of demonstrations needed to obtain a variance to closed-cycle cooling (Ref. 2). Public Law 92-500, Section 316(a) provides that "if an owner or operator after opportunity for a public hearing can demonstrate to the satisfaction of the relevant permitting authority that any effluent limitation proposed for the control of the thermal component of the discharge from a power plant will require effluent limitations more stringent than necessary to ensure the protection and propagation of a balanced, indigenous population of shellfish, fish, and wildlife in and on the body of water into which the discharge is made, the permitting authority may impose a less stringent, alternative effluent limitation." The burden of the proof is on the discharger to demonstrate that an alternative effluent limitation will indeed protect the biological community. Rules and regulations governing Section 316(a) published in the *Federal Register* set forth three types of demonstrations by which the discharger can submit evidence for a less stringent effluent limitation.

Type 1: The use of on-site data to demonstrate absence of prior appreciable harm to the biological community.

Type 2: A demonstration using EPA's draft water-temperature criteria that conditions at the site would protect important representative species.

Type 3: The use of a combination of biological and engineering data to demonstrate no adverse effects.

It seems reasonable to extend this law to include effluent discharges other than thermal if the same basis for variance is also required. This might be a savings to the discharger, would provide evidence of the effect, if any, on the biological community, and increase the information base about the acceptable range of discharges to aquatic communities.

PREDICTIVE AND REACTIVE CONTROL

There is no probe devised by man that will measure toxicity! Living organisms, unlike any other analytical tool available, will respond to every possible substance or mixture of substances at some concentration, no matter what its chemical or physical characteristics may be. The ability of living organisms to integrate information about the environment they inhabit is unequalled among the instruments devised by man. As a consequence, if one wishes to know the effects of a metal or a chemical substance on living organisms, the effect must be determined by a direct measurement made upon the organisms themselves. The environmental conditions should be those at which exposure is likely to occur since the same species will respond differently to the same concentration of a toxicant if the environmental conditions under which the two exposures occur are not identifical. As a consequence, precise determinations of response thresholds must be made on a site-specific basis! Similarly, different species of organisms under identical environmental conditions will respond differently to the same concentration of the chemical so it is advisable to use indigenous organisms when possible. A biological response alone will not be particularly useful since, without knowing the concentrations at which exposure occurs or the environmental conditions of pH, temperature, water quality, and so on, the predictive value of the biological data gathered will be minimal.

In the United States and many other industrialized countries, legislation is appearing calling for premanufacture and marketing testing of all chemicals. In the United States, the Toxic Substances Control Act (TSCA;Public Law 94-469-October 11, 1976) provides that no person may manufacture a new chemical substance or manufacture or process a chemical substance for a new use without obtaining clearance from the U.S. Environmental Protection Agency. One of the

main objectives of TSCA is to establish a procedure for evaluating hazard to human health and the environment before widespread use of a new chemical occurs. After examining the data produced to implement the evaluation, the administrator of EPA must judge the degree of risk associated with the extraction, manufacturing, distribution in commerce, processing, use, and disposal of the chemical substance. If the chemical presents an unreasonable risk of injury to health or the environment, the administrator of EPA may restrict its use or ban it entirely. Such preuse hazard evaluations are presently almost entirely based on single species toxicity tests. Generally, fish, diatoms, or invertebrates are placed in containers and exposed to a graded series of concentrations of the chemical or effluent being tested, and the degree of lethality or other adverse biological responses associated with this exposure is determined. Because many of these chemicals transform rapidly, present practice is to use continuous flow exposure rather than batch testing. However, even this additional requirement produces a test situation with some limitations. For example, no other species such as predators or prey are available for interaction, and the environment is necessarily simple and quite unlike a natural one. In a like manner, toxicity tests for materials to which humans are likely to be exposed are also simplistic and not fully reflective of "real world" conditions. Despite the substantial number of theoretical drawbacks, such tests have worked remarkably well. In part, this may be because laboratory conditions, however skillfully maintained, are less suitable to the organisms being tested than the natural environment. As a consequence, the response threshold is likely to be lowered rather than increased, adding a safety factor to the determinations. Only a few species can be tested under laboratory conditions both because of economic considerations and because there is only sufficient information on a relatively few species to enable scientists to culture them under laboratory conditions. Additionally, natural environments are notoriously variable, particularly when episodic events such as 20 year floods are concerned. A generous budget most likely would not give the laboratory investigators determining the toxicity of an effluent or chemical substance the capability to test it under all possible conditions likely to occur and against all the species likely to be exposed. In short, despite the remarkable usefulness of the laboratory tests, one needs to know whether the response in natural systems to a discharge is less than, greater than, or comparable to that predicted. Thus, one's responsibility in determining the biological effects of a chemical discharged into an environment does not end with the laboratory testing but must be supplemented with field testing once actual discharges begin in order to determine whether or not the calculations of no adverse biological effects concentrations were sound. In short, any quality control system including effluent discharges, based on effect-related biological criteria, must have a feedback loop from the environment to determine whether the response was as predicted. This relationship is depicted in Fig. 1.

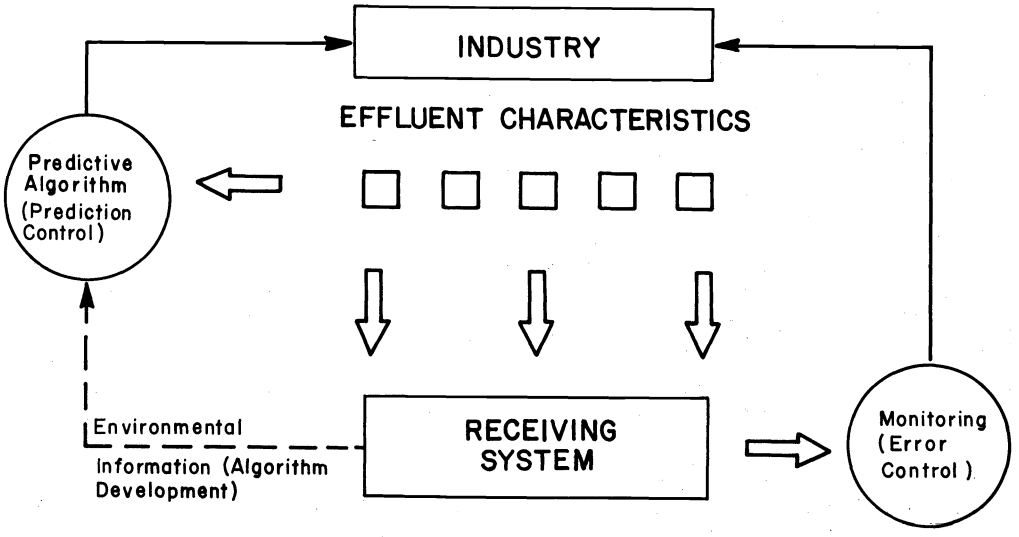


Fig. 1. Information flow in environmental control processes (Ref. 3).

PREDICTIVE CONTROL

The process of hazard evaluation

The discussion noted earlier that the process of hazard assessment requires scientific judgment based on evidence, and, obviously, for the judgment to be sound the evidence must be accurate and systematically gathered for the particular purpose of estimating the probability of harm to living organisms. Much of the evidence used for this purpose in industrialized countries

has been produced for entirely different purposes. Since most of it has been generated by the academic community, it consists of theses, dissertations, and research investigations carried out for various academic reasons. Only a small fraction of the total information base now available was most likely generated for the purpose of hazard evaluation. Nevertheless, much is useful and is practically all we have available at present. Consequently, it must be used with a recognition of its limitations. However, a truly sound scientifically justifiable estimation of hazard must be based on evidence generated for this purpose. How much of this information should one expect? First of all, given the millions of chemicals already known and the thousands produced annually in industrialized countries, how does one determine which of the chemicals deserve high testing priority? In a short but fascinating article entitled "Chemicals:How Many Are There?" Thomas H. Maugh II (4) gives a one-page summary of the overall problem. The American Chemical Society's computer registry contains 4,039,907 distinct entities, and the number has been increasing at a rate of about 6,000 per week. Of these, the ACS has given EPA a preliminary list of approximately 33,000 chemicals that are thought to be in common use. EPA believes there may be as many as 50,000 chemicals in daily use not including pesticides, pharmaceuticals, or food additives. The Food and Drug Administration estimates approximately 4,000 active ingredients in drugs and 2,000 more used as excipients, as well as 2,500 additives for nutritive purposes and 3000 more to promote product life. Taking all of these together, Maugh estimated about 63,000 chemicals in common use.

The number of people competent to carry out toxicity tests and environmental fate-and-effects determinations, however, is exceedingly small (Ref. 5). Although people can be quickly trained (i.e., a year or two) for the crude short-term tests using lethality as an endpoint, it is extremely time-consuming to educate people to conduct the long-term tests or interpret the data. Moreover, facilities suitable for carrying out such tests are not abundant, and funds for conducting hazard evaluations are limited even when facilities and personnel are available. One cannot carry out every test developed by the academic community on every chemical. Recognition of this simple fact makes mandatory the development of a process for determining testing priority. A schematic for doing so from *Principles for Evaluating Chemicals in the Environment* (Ref. 6) is shown here in Table 1.

TABLE 1. Scheme for classification of chemicals according to biological impact and dispersal

Chemical Disperal	Biological Impact*		
	High (1)	Medium (2)	Low (3)
(1) Widespread, high release	1	2	3
(2) Widespread, low release	2	4	6
(3) Localized, high release	3	6	9
(4) Localized, low release	4	8	12

*Low number indicates high testing priority.

Once priorities have been set and testing begun, some guidelines must be set for determining when one has sufficient information to make a judgment on the probability of harm. Such a decision depends on the relationship between two different kinds of information: (a) the environmental concentration of the chemical, and (b) the concentration below which no adverse biological effects are produced. At the beginning of any testing sequence, confidence in one's ability to determine the concentration producing no adverse biological effects is rather low. This is because only a relatively few species have been tested under a relatively limited array of conditions. As more and more information becomes available, one's confidence increases that the no adverse effects concentration threshold has been reasonably well determined. It is, of course, a *sine qua non* that one can never determine the no adverse biological effects concentration with absolute certainty. This is also true, of course, for the environmental concentration. However, during an environmental hazard evaluation procedure, increasingly accurate estimates are obtained about: (a) the concentrations of the chemical that do not cause adverse biological effects, and (b) the environmental concentrations that will result from production and use of the chemical. When one has reduced uncertainty to the point where one can be reasonably certain that: (a) the two concentrations are indeed distinct (i.e., the confidence bands do not overlap), and (b) the environmental concentration is lower than the adverse effects concentration, there may be justification for terminating testing and concluding, *at a certain risk level*, that introduction of the chemical will not cause an environmental hazard. As in all other situations, risk can never be reduced to zero. This situation is depicted graphically in Fig. 2.

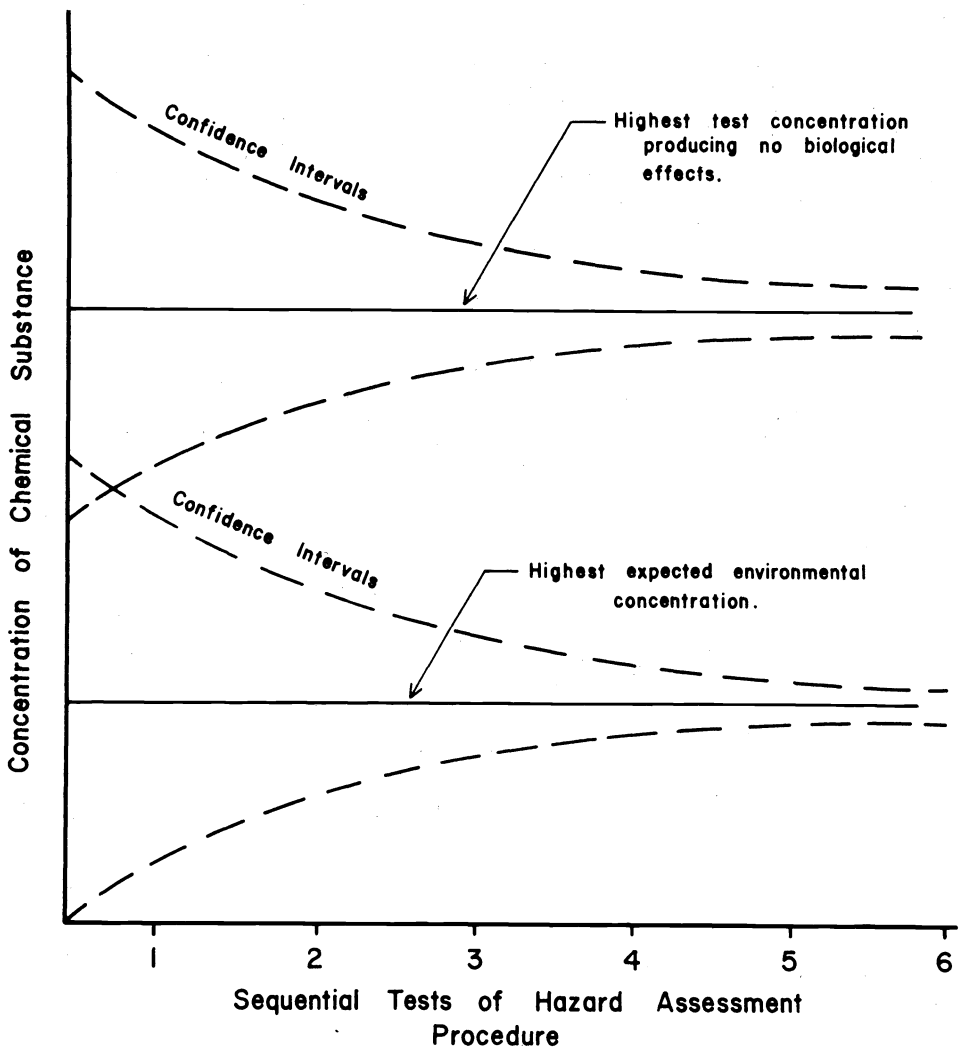


Fig. 2. Diagrammatic representation of a sequential hazard assessment procedure demonstrating increasingly narrow confidence limits for estimates of no biological effects concentration and actual expected environmental concentration (Ref. 7).

Verification of predictions

As already mentioned, the development of predictive models for hazard evaluation is a relatively new exercise. The means of verifying these estimates of no adverse effects concentrations is also new. At the present time, standard methods for providing estimates of toxicity are based on single species exposure in artificial containers. The problem of extrapolation to "real world" conditions has already been mentioned. Verification of the reliability of predictions before actually discharging the effluents or allowing the chemical to intrude into the environment would be welcome. This verification should be accomplished with systems of greater complexity (i.e., more species and more complex environmental conditions) than those existing in the toxicity tests now accepted as standard methods. A promising means of verifying these predictions is available with the microcosms being developed by Metcalf (8) and a number of other investigators (Ref. 9). These microcosms (larger scale units are sometimes called mesocosms) afford an opportunity to make some corrections in the estimates if necessary. Space does not permit a discussion of these, but a search of the literature will reveal a number of types. A relatively recent reference containing some literature on this subject is Cairns et al. (7). The utility of model ecosystems or microcosms for this purpose remains to be proven. However, the possibility of reducing the number of major environmental catastrophies by testing chemicals determined to be possibly hazardous under conditions of proposed use justifies the research and development of such systems and the generation of data necessary to determine their utility. Mistakes made in model ecosystems, however expensive, are clearly a minute fraction of costs of making mistakes in natural systems. Kepone in the James River in the State of Virginia is estimated to require over 100 years to degrade naturally and removing it artifically could cost as much as seven billion dollars.

PROTOCOLS FOR EVALUATING THE EFFECT OF CHEMICAL SUBSTANCES ON AQUATIC LIFE

Reference has already been made to the need for systematically gathering data in order to make a scientifically justifiable evaluation of hazard. Generally, the organized system for doing so is called a protocol. One of the earliest of these protocols was developed by Cairns and Dickson (10) in 1973 for the U.S. Army Medical Research and Development Command to give the army guidance in generating the necessary data base for developing water quality standards for specific munition industry chemicals. However, the guidelines are so broad that they would fit almost any chemical substance.

There are a number of basic goals in the establishment of a protocol in terms of information content:

1. To determine the range of variability of species from different trophic levels in natural systems.
2. To determine the extent to which environmental quality affects the expression of toxicity.
3. To determine which of the species tested is the most sensitive and for that species which is the most sensitive life history stage (i.e., the weakest link). The assumption is made that by protecting the most sensitive species the others will be protected as well.
4. To use short-term inexpensive screening tests on a wide variety of species to determine which are the most sensitive and the concentration at which critical responses occur before embarking on the expensive and time consuming long-term tests. As a consequence, the most appropriate concentrations can be used for the long-term tests rather than a broad spectrum of concentrations, and a few select highly sensitive species can be used instead of a broad array of species. In short, information gathering on the most vulnerable components of the ecosystem is enhanced.
5. To gather sufficient information so that additional tests are not carried out beyond the short-term tests if information shows that production and marketing of the chemical or discharge of the effluent can proceed at a reasonable level of risk.

One should remember that only a relatively few species of the many thousands existing in nature can be tested. Therefore, even the test protocols with an array of species and an array of conditions still only give some general notion of the spectrum of responses from which a projection must be made using scientific judgment based on total accumulated knowledge of toxicity.

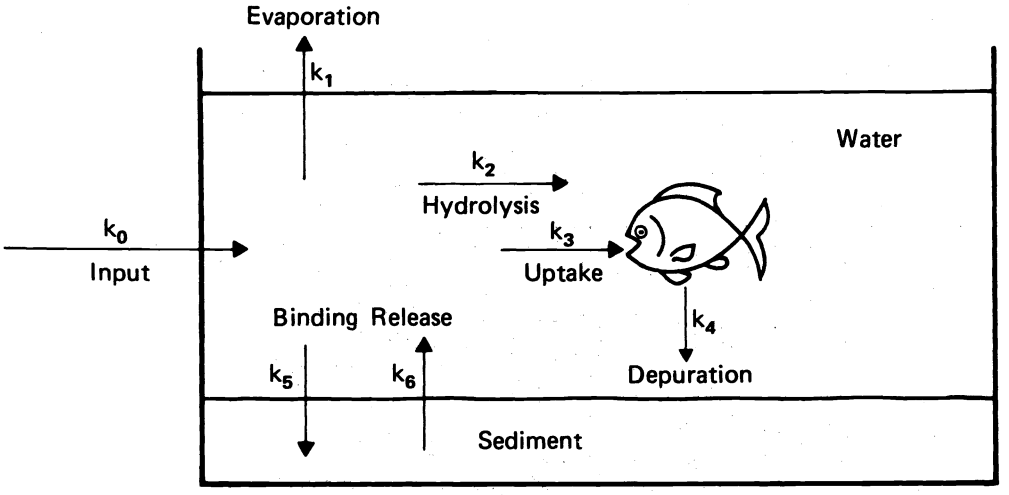
Another feature of the Cairns and Dickson (10) protocols that is common to many others is the alteration of data gathering and decision making. Generally, a tiered or phased approach is used (there are three tiers in the Cairns and Dickson protocols) in which a body of evidence is gathered and then certain questions asked so that additional data may not be gathered without scientific justification for doing so. The reasons for gathering the data must be explicitly stated, and the way in which the new data will contribute to the decision making process must be clear. Another feature of the protocol just discussed is the requirement that laboratory and field tests be carried out simultaneously and that the predictions of one be checked with the other. This, of course, can only be done when a plant is already operating. For cases where no actual manufacturing process is underway, verification must be carried out in model ecosystems or in small sections of natural systems designated for this purpose. However it is done, some field verification of the accuracy of the predictions made on the basis of the laboratory protocol should be mandatory. Additional protocols are found in Cairns et al. (7), but the most complete set representing a large number of industrialized countries may be found in Dickson et al. (11). Present standard methods are usually tier 1 or screening test level, but there are a few tier 2 tests. The parameters to be used and the methods to be used for long-term tier 3 tests have not yet, to the best of my knowledge, been formally and officially endorsed as standard methods. There is great need for this to be carried out as expeditiously as possible and, more important, for systems level effects as opposed to species level effects to receive greater attention. Systems level effects will be discussed later in this paper.

ENVIRONMENTAL PATHWAY DETERMINATION

One must be concerned not only with the effect of a chemical upon the environment, but also with the effect of the environment on the chemical. Various types of chemical, physical, and biological transformations are common for most chemicals. Generally, the transformation results in less toxic products. As a consequence of transformation, dilution, and other factors, the diminution of toxicological properties may be rather substantial. Even for persistent chemicals, we must know the environmental pathways; how the material is partitioned between air, water, and soil; what types of environmental sinks are operative; and under what conditions releases might be expected from these sinks. For example, a chemical associated with lake sediments requires a different assessment strategy from that for a chemical associated with the water column: toxicity tests for the former should involve benthic organisms, whereas planktonic organisms would be most suitable for the latter. A rapidly degrading substance would require fewer chronic tests than a persistent one.

Stern and Walker (12) have developed an approach to identify the principal medium into which a chemical may be distributed after release into the environment. The following series of tests was used: water solubility, partition coefficient (octanol/water), adsorption by natural solids, desorption or leaching, and volatility. For example, if a chemical is soluble in water, does not transfer to octanol, does not readily adsorb to soils, readily leaches from areas in which it is deposited, and has a low degree of volatility, testing of persistence and ecological effects could be limited to these conditions and biological targets associated with the liquid phase of bodies of water and, to a lesser degree, their sediments. The integration of data developed in tests of environmental mobility with the information required by TSCA Section 8 will provide a useful indication of possible "target" organisms.

The new environmental rates approach (Ref. 13) requires that properties be measured as time-concentration rates, which are then incorporated into a suitable model for predicting environmental concentrations. Figure 3 depicts a pond model, the key properties, and the materials-balance equation for predicting the fate of chloropyrifos in the pond. The predicted



MATERIAL BALANCE EQUATION

$$V\frac{dCw}{dt} = k_0 - k_1 ACw - k_2 VCw - k_3 FCw + k_4 FC_f + k_5 SCw + k_6 SC_S$$

Input — Evaporation — Degradation (Hydrolysis) — Fish Uptake — Fish Depuration — Sediment Binding — Sediment Release

Where V = Volume of water, ml
A = Surface area, cm^2
F = Fish mass, gm
S = Sediment mass, gm
Cw = Concentration of chemical in water
k = Rate constant
C_f = Concentrations of chemical in fish
C_S = Concentration of chemical in sediment

Fig. 3. Pond model (Ref. 13).

and experimentally found concentrations in the fish and water (Table 2) were in close agreement.

TABLE 2. Predicting the fate of chlorpyrifos in a pond* (Ref. 13)

Compartment	% of total	
	2 days	25 days
Water	48.0	0.8
Soil	25.0	0.5
Fish	0.8	<0.1
Air	3.8	11.4
Metabolized	2.9	11.0
Hydrolyzed	25.0	76.0

*Seven-day (similar agreement at days 2, 4, and 28) concentrations of chlorpyrifos: (a) water (5.75 µg/liter at t=0)-predicted 1.0 µg/liter (Ref. 14); found 1.0 µg/liter (Ref. 15); (b) fish--predicted 0.8 µg/gram (Ref. 14); found 1.0 µg/gram (Ref. 15).

The environmental concentration of a chemical is governed by the properties of the chemical, the rate of its introduction into the environment, and the characteristics of that specific environment. Many of the properties - e.g., molecular structure, water solubility, vapor pressure, absorption spectra (ultraviolet and visible), and particle size (if the substance is particulate) - may be readily available from data banks. Others that may be readily attainable before biological program design begins could include some rate constants - i.e., for photodegradation (ultraviolet and visible), biological degradation, chemical degradation, evaporation, sediment binding, uptake by organisms, depuration by organisms - and some partition coefficients - i.e., octanol/water, air/water, and sediment/water. The characteristics or properties of the environment-e.g., surface area, depth, pH, flow/turbulence, carbon in sediment, temperature, salinity, suspended sediment concentration, trophic status, and absorption spectra (ultraviolet and visible)-are equally fundamental (Ref. 16).

EFFECTS:SPECIES VERSUS ECOSYSTEM

The literature abounds with laboratory tests on the effects of chemicals on species. A variety of responses have been studied - an illustrative selection follows: death; fecundity; growth rate; swimming speed; visual acuity; behavior; reproduction; susceptibility to parasites; feeding rate; predation; and equilibrium. However, as the degree of complexity of the system being studied increases (i.e., from species [population] to community or ecosystem), new properties emerge that were not apparent at the lower levels of organization (e.g., energy flow). Unfortunately, the higher the level of organization being studied the fewer the tests that are available to do so and the more expensive such tests become. However, there is no direct evidence that toxicity tests on species alone will enable one to accurately estimate concentrations that will not cause harm to communities and ecosystems. Regulatory agencies will benefit from development of both short- and long-term testing procedures for estimating hazard to communities and ecosystems from various environmental concentrations of chemicals. These should provide data suitable for evaluating the hazards presented to an ecosystem exposed to known chemical substances in reasonably well defined concentrations. In order to develop suitable evaluation schemes (or protocols), one must define appropriate ecological parameters indicating stress, determine which tests would best measure changes in these parameters, and develop ways to assemble a data base for a reliable assessment of the potential hazards to an ecosystem. Although laboratory test schemes involving microcosms and mesocosms (systems involving multiple species) are generally most cost effective and also reduce environmental contamination, some field work will be essential to verify the estimates based on laboratory results. No evaluation scheme could be considered an adequate representation for assessing ecosystem damage without field verification. In the development of systems to test community and ecosystem responses, intermediate new facilities (e.g., "closed" systems, "greenhouse" testing) should be considered to bridge the present gap between laboratory single species toxicity tests and field tests on communities and ecosystems.

Why single species tests alone may be insufficient

There are a number of reasons why single species tests may be inadequate for protecting communities and ecosystems. Some of the more important reasons follow: (a) The community response determines the effective concentration of the chemical substance if we wish to

protect a large array of species instead of a select few. (b) Metabolic processes in the community often transform the substance into something less hazardous (or more), and this biotransformation is impaired if the community is damaged. (c) A direct single species impact may be overridden by other impacts at the community or ecosystem level. (d) Adjustments may be possible at the community or ecosystem level that are not possible at the species level.

Some excellent evidence of the problems associated with single species evaluations and the value of ecosystem level evaluations may be found in case histories of pesticides, radionuclides, and heavy metals.

Ecosystem properties that may provide effect-related effluent criteria

There are a number of system properties on which effect-related effluent criteria for pollutants might be based. These contrast strongly with the parameters used to determine effects on species. In both cases, however, one has more confidence in estimates based on multiple parameters than on one. An illustrative list of system properties follows: diversity, evenness, productivity, nutrient and energy transfer, biomass, decomposition, connectivity, stability properties (elasticity and inertia), density dependent composition, sensitivity, and geographic specificity. Unfortunately, standard methods for making these determinations are not now available although a variety of methods are in use. One hopes ecologists will give high priority to production and formal professional endorsement of standard methods for assessing systems effects. Until that happens, effect-related effluent criteria for pollutants will be based primarily on single species toxicity tests.

CONCLUSIONS

Most of the current pollution problems would have been significantly reduced if present effect-related effluent criteria had been properly used. Despite some theoretical weaknesses already discussed, toxicity tests have proven to have great utility. Although more research is needed, there is no justification for delaying widespread use of methods with a long history of effective use. A variety of such methods are listed in articles in the reference section of this article (Ref. 3-18). Sound management practices will be effective only if engineers, chemists, biologists, economists, and a variety of other special interest groups work well together to provide a solution to industrial pollution.

For most people in industrialized societies, awareness of environmental pollution began in the 1960; quantification of the damage was carried out in the 1970s. Development of prediction models has already begun and may well dominate pollution abatement research for the next decade. If this effort is as successful as early evidence suggests, the last decade of the century can be devoted to the employment of the quality control practices already developed in a series of management programs. If this all transpires, society may enjoy technological and natural benefits simultaneously with significantly reduced risks.

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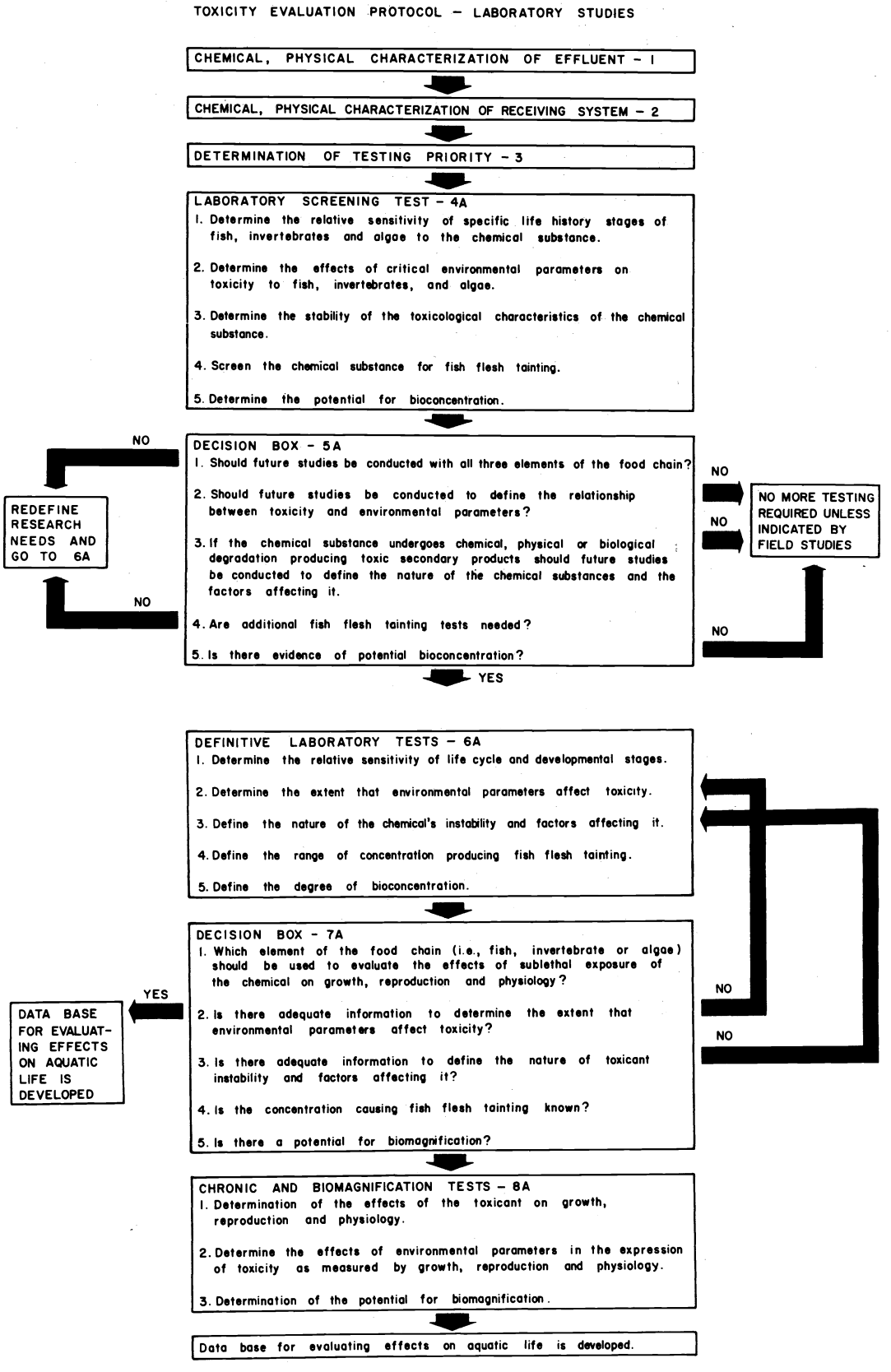
ADDENDUM

The field and laboratory protocols mentioned in the text are reproduced with permission of the American Society for Testing and Materials, 1916 Race Street, Philadelphia, Pennsylvania. The text accompanying the protocols may be found in Cairns and Dickson (10).


TOXICITY EVALUATION PROTOCOL – LABORATORY STUDIES
CHEMICAL, PHYSICAL CHARACTERIZATION OF EFFLUENT – 1
CHEMICAL, PHYSICAL CHARACTERIZATION OF RECEIVING SYSTEM – 2
DETERMINATION OF TESTING PRIORITY – 3
LABORATORY SCREENING TEST – 4A
1. Determine the relative sensitivity of specific life history stages of fish, invertebrates and algae to the chemical substance.
2. Determine the effects of critical environmental parameters on toxicity to fish, invertebrates, and algae.
3. Determine the stability of the toxicological characteristics of the chemical substance.
4. Screen the chemical substance for fish flesh tainting.
5. Determine the potential for bioconcentration.
DECISION BOX – 5A
1. Should future studies be conducted with all three elements of the food chain?
2. Should future studies be conducted to define the relationship between toxicity and environmental parameters?
3. If the chemical substance undergoes chemical, physical or biological degradation producing toxic secondary products should future studies be conducted to define the nature of the chemical substances and the factors affecting it.
4. Are additional fish flesh tainting tests needed?
5. Is there evidence of potential bioconcentration?
NO
REDEFINE RESEARCH NEEDS AND GO TO 6A
NO
NO
NO MORE TESTING REQUIRED UNLESS INDICATED BY FIELD STUDIES
NO
YES
DEFINITIVE LABORATORY TESTS – 6A
1. Determine the relative sensitivity of life cycle and developmental stages.
2. Determine the extent that environmental parameters affect toxicity.
3. Define the nature of the chemical's instability and factors affecting it.
4. Define the range of concentration producing fish flesh tainting.
5. Define the degree of bioconcentration.
DECISION BOX – 7A
1. Which element of the food chain (i.e., fish, invertebrate or algae) should be used to evaluate the effects of sublethal exposure of the chemical on growth, reproduction and physiology?
2. Is there adequate information to determine the extent that environmental parameters affect toxicity?
3. Is there adequate information to define the nature of toxicant instability and factors affecting it?
4. Is the concentration causing fish flesh tainting known?
5. Is there a potential for biomagnification?
YES
DATA BASE FOR EVALUATING EFFECTS ON AQUATIC LIFE IS DEVELOPED
NO
NO
CHRONIC AND BIOMAGNIFICATION TESTS – 8A
1. Determination of the effects of the toxicant on growth, reproduction and physiology.
2. Determine the effects of environmental parameters in the expression of toxicity as measured by growth, reproduction and physiology.
3. Determination of the potential for biomagnification.
Data base for evaluating effects on aquatic life is developed.

TOXICITY EVALUATION PROTOCOL - FIELD STUDIES