UNITED STATES ENVIRONMENTAL PROTECTION AGENCY·

WASHINGTON. D.C. 20460

October 13, 1983

Mr. William D. Ruckelshaus
Administrator
401 M Street, SW
Washington, D.C.   20460

Dear Mr. Ruckelshaus:

The Science Advisory Board has endorsed the enclosed report of its
Environmental Effects Transport and Fate Committee, entitled "Report on Site
Specific Water Quality Criteria". The Committee has carried out an indepth
review of this issue, and its major conclusions and recommendations are discussed
in the enclosed report.

The Science Advisory Board strongly encourages the Office of Water Regulations
and Standards to incorporate the recommendations in the report into its Site
Specific Water Quality Standard Guidelines. We believe that the site specific
guidelines and the accompanying recommendations offer a new landmark in approaches
to the development of standards to assure the quality of the aquatic environment.

The Science Advisory Board wishes to thank the dedicated Agency staff who
provided great assistance in the preparation of this report. The Board also
urges further consultation and involvement by its Environmental Transport and
Fate Committee in the further development of the guidance.

Sincerely yours,

Rolf Hartung, Chairman
Environmental Effects, Transport
and Fate Committee

Earnest F. Gloyna, Chairman
Executive Committee

REPORT ON

SITE-SPECIFIC WATER QUALITY CRITERIA

by

The Environmental Effects, Transport, and Fate Committee

of the

Science Advisory Board

October 13, 1983

EPA NOTICE

This report has been written as a part of the activities
of the Agency's Science Advisory Board, a public advisory
group providing extramural scientific information to the
Administrator and other officials of the Environmental
Protection Agency. The Board is structured to provide a
balanced expert assessment of scientific matters related
to problems facing the Agency. The contents of this
report do not necessarily represent the views and policies
of the Environmental Protection Agency.

TABLE OF CONTENTS

ENVIRONMENTAL EFFECTS, TRANSPORT AND FATE COMMITTEE

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CHARGE TO THE COMMITTEE

On November 4, 1982, the Environmental Effects, Transport and Fate Committee of the Science Advisory Board accepted the charge to evaluate the scientific validity of the procedures proposed as guidance for the States by the U. S. Environmental Protection Agency for the development of site-specific water quality standards.

Among the issues that the Committee was requested to address were the following:

1. Determination of whether or not the site-specific guidelines more correctly protect the various uses of aquatic life by accounting for toxicological differences in species sensitivity or water quality at specific sites for designated uses.

2. Evaluation of species sensitivity ranking and toxicological effects derived from appropriate laboratory tests.

3. Discussion of the stringency of site-specific criteria developed from biological data on aquatic or terrestrial animals vs. the concentrations of pollutants affecting plants or concentration/effect data in the category "Other Data" found in the national criteria documents.

4. Evaluation of procedures to modify criteria to account for some characteristics of local sites. Among these procedures are the Recalculation Procedure, the Indicator Species Procedure, the Resident Species Procedure, the Heavy Metal Speciation Procedure, the Historical Procedure, and the Final Residue Value Procedure.

In the course of its deliberations, the Committee was also requested to evaluate a series of site-specific test studies which had been conducted according to the draft guidance procedures.

EXECUTIVE SUMMARY

In November 1982, the Environmental Effects, Transport and Fate Committee of the Science Advisory Board was requested by the Assistant Administrator for Water to review a set of proposed guidelines by which national water quality criteria could be adapted to derive local water quality standards, taking site-specific conditions into account.

The Committee determined that the basic goal, to derive site-specific standards, was both important and necessary. The application of single national criteria in the form of local standards to situations as diverse as the Great Lakes, the lower Mississippi, or a Colorado mountain stream had been challenged frequently before.

Since the proposed guidelines were often modifications of the existing methodology for the setting of national water quality criteria, many aspects involving those methods also had to be considered.

The Committee found that many aspects of the proposed guidelines did not make adequate use of existing information and that the logical foundation of some sections of the guidelines was flawed.

The site-specific environmental problem, as presented to this Committee, was poorly specified. As a result, the existing procedures are inadequate and in need of revision.

While the Clean Water Act specifies that the physical, chemical, and biological integrity of the environment shall be protected, the Agency has failed to provide more specific benchmarks to serve as the basis for designing protective strategies and against which the performances of various protective strategies could be judged.

The proposed guidelines, as well as the national water quality criteria, are largely based on laboratory toxicity studies. Aside from issues related to the statistical validity related to the desired protection of 95% of the species or 95% of the families, laboratory toxicity tests fail to account for inter-actions between species, ecosystem level effects, interactions with other chemicals, and modifications by local water quality characteristics. The species tested in the laboratory are assumed to reflect significant or important species in the environment.

The Committee concluded that the sum of such assumptions made it essentially impossible to discern a logical framework

which would guarantee achieving the protection of the environmental integrity of aquatic systems. However, the Agency did recognize the need to account for the effects of local water quality on the toxicity of pollutants; did make progress in selecting representative species; and did recognize the need for field verification as part of the proposed site-specific guidelines. However, the extent to which the national criteria, or by implication the proposed site-specific guidelines, provide protection appears to be mainly based on their conservative features, rather than on any defensible scientific derivation.

While the national criteria and the site-specific modifications are centered around laboratory toxicity studies, the Committee strongly recommends that the site-specific standards should be centered around the responses of ecosystems and their components in an environmental rather than laboratory setting.

Thus, the Committee places a much greater emphasis than does EPA on biological, chemical and physical monitoring. The monitoring must be relevant to the detection of adverse impacts, and it must view the ecosystem in dynamic terms dedicated to specific uses.

The proposed guidelines suggested that site specificity could be achieved through the application of one or more procedures:

1. A "recalculation procedure," which recalculates the criteria by taking account of differences between the species tested as part of the national data base and those species which actually occurred or were expected to occur at a local site. This procedure is basically an extension of the methods for the derivation of the national criteria based on laboratory toxicity testing. While it does represent a logical refinement of the national criteria, it is subject to the same shortcomings, with respect to ecosystem applicability, as the national criteria.

2. An "indicator species procedure," in which acute bioassays are conducted in site water and in defined laboratory water to derive a ratio which represents the impact of local water quality. This ratio is then applied to the national data base. The procedure assumes that the ratio between acute and chronic toxicity is constant for a given chemical and that the influences of water quality on toxicity in short-term tests at high doses will also hold for long-term low level exposures. Preliminary studies have indicated that this water effects ratio may differ significantly depending upon the species selected,

especially between fish and invertebrates. In
addition, such a water effects ratio appears to be
highly dependent upon the sampling and storage methods
for the local site water.

3. A "resident species procedure," in which the national
toxicity data base is repeated for both acute and
chronic toxicity using local species in site water.
This method still retains many of the shortcomings of
the toxicity-based national criteria, except that it
may account for the impact of local water quality better
than the "indicator species procedure" does.

4. A "heavy metals speciation procedure," which depends
upon the metal concentration in a 0.45 μm filtrate,
rather than total metal concentration. The Committee
agreed that the chemical and physical forms of heavy
metals in water were important determinants of biological
activity. However, better methods than simple filtration
are available for many forms and should be utilized.

5. A "final residue value procedure," which is essentially
identical to that in the national water quality criteria
methodology and is based on laboratory data. Such
laboratory data have, in practice, often differed from
field data. The final residue values for site-specific
conditions should be coupled more closely to actual
field conditions.

As part of its endeavors relating to the site-specific
guidelines, the Agency commissioned several site-specific
evaluations of the proposed guidelines. In the aggregate,
these studies were judged to be very inadequate, but they
provided a useful learning experience; they tended to point
out some of the frailties of the originally proposed methodology,
a number of which are also cited in this report. Evaluation
of the studies also indicated that their design needs to be
improved and that the Agency needs to develop better technical
guidance for such studies. This report makes a number of
recommendations about how that might be accomplished.

Though the Committee was often critical of various aspects of
the proposed guidelines, such criticisms were offered with
the intent to be helpful. Above all,

- the Committee urges the Agency to continue its
development of site-specific guidelines and not to
abandon such efforts because of initial criticisms.

INTRODUCTION

Section 304(a) of the Clean Water Act specifies that the Administrator of the Environmental Protection Agency promulgate criteria, based on the latest scientific knowledge, to protect both the biological integrity and specified uses of the Nation's waters. EPA to date has developed national criteria for a number of pollutants and is currently developing guidance for site-specific standards. In this report the scientific basis of the site-specific standards is examined.

For many reasons, national water quality criteria are often deficient when applied as standards, without modification, to local conditions. Thus, developing scientifically defensible site-specific water quality standards is an extremely important and necessary task.

The proposed methodology for the derivation of site-specific standards is inseparably intertwined with the methodology for setting national water quality criteria. Therefore, it was impossible to appropriately consider the site-specific methodology without also examining many aspects of the methodology for deriving the national criteria.

The national water quality criteria documents constitute a valuable collection of background information on the effects of pollutants on selected species under laboratory conditions. Unfortunately, the national criteria did not adequately consider the broad range of interactions between pollutants and among species at the ecosystem level, which becomes very important when standards are to be set to appropriately protect an ecosystem at any given site.

The Environmental Effects, Transport and Fate Committee of the Science Advisory Board identified a number of important guidelines and the national water quality criteria from which those guidelines are, in large part, derived. The following sections of this report address those problems and suggest approaches to remedy them.

While the Committee has found serious deficiencies in the scientific bases for the criteria and the standards, it strongly urges the Agency to develop those scientific bases and incorporate them into its methodologies as rapidly as possible.

In various sections of this report, a series of demonstration projects, which were conducted to explore site-specific guidelines, is referred to. There are separate brief critiques of these in Appendix 2.

MAJOR RECOMMENDATIONS

1. The role of environmental monitoring as a tool in the development of site specific water quality standards needs to be greatly expanded.

2. The level of desired protection, commensurate with the protection of environmental integrity, needs to be carefully specified so that the performance of various protective strategies can be evaluated.

3. Laboratory toxicity tests are useful for the derivation of national water quality criteria, but since criteria derived from such tests alone cannot consider local ecological interactions, they should not be converted into site-specific standards without studying effects at the ecosystem level.

4. The statistical procedures for the site-specific criteria cannot be substantially improved in the absence of local data by which relevant ecological dynamic states may be defined, measured, and modeled. For this reason, we do not recommend that the basic calculation procedure be modified.

5. The "indicator species procedure" is based on a number of assumptions which are presently unverified. This procedure has often generated contradictory results in a series of test cases. We recommended that this procedure not be used until further research has demonstrated its applicability.

6. The proposed filtration method for the speciation of heavy metals is inadequate. The Agency should determine the most appropriate method for estimating the biologically active forms of each individual pollutant.

7. The Agency should complete development of the "historical procedure," including procedures for required monitoring.

8. Actual residue concentrations and their environmental dynamics should be given precedence over laboratory-derived bioconcentration coefficients in the derivation of site-specific final residue values.

9. The Committee urges the Agency to continue its development of site-specific guidelines and not to abandon such efforts because of initial criticisms.

PROBLEM STATEMENT AND RESOLUTION

The problem of environmental protection against toxic substances in the aquatic environment is poorly specified.

Limitation of Toxicity Testing

P. L. 92-500 mandates that the integrity of the environment be protected. Yet, water quality criteria and the proposed site-specific standards are largely based on acute and chronic toxicity testing of single chemicals on single species in laboratory settings. Interactions between exposed species, between chemicals in the form of potentiation or antagonism, and of chemicals with ecosystem properties are ignored in the proposed methodology used to set site-specific standards. Therefore, current methodology can be appropriate in concept if and only if all of the following parameters are either satisfied or produce a negligible effect:

1. The appropriate species have been tested based on ecological importance and human uses, particularly consumption, taking into account toxic residues.

2. The exposure is environmentally realistic in terms of physical and chemical identity.

3. The stresses under laboratory conditions not related to the toxicant are equivalent to the non-specific stresses found in the environment.

4. Interactions with other components in the environment produce insignificant effects.

5. The toxicant has no effect on interactions between species, such as competition, predation, commensalism, or parasitism.

6. The toxicant produces no other effects on the ecosystem that may affect the test species indirectly.

It seems unlikely that all of these conditions will be met in the vast majority of cases in which toxicants are tested in the laboratory and the results are subsequently applied directly to natural systems.

Thus, while single species toxicity testing in the laboratory provides important information, e.g., relative toxicity of compounds or relative sensitivity of species under specific laboratory conditions, that information is incomplete for setting appropriate standards for the protection of the environment as mandated by P.L. 92-500.

Standards, which are exclusively based on laboratory toxicity testing, would therefore be accurate not by design, but by chance. At times such standards appear to be overly pro- tective, but at other times they fail to protect adequately. The reasons why the standards appear to protect more often than not may result from the various conservative approaches built into the criteria and standard-setting procedures.

Role of Toxicity Data

Bioassays are generally performed on a few selected life stages of a selected number of species under controlled laboratory conditions. Response dynamics are very sensitive to the experimental conditions; and, therefore, great care must be taken to manipulate only those variables which are subject to testing. Inadvertent variations in nontest variables will contribute spurious components to the observed behavior, masking effects of the intended treatment.

- Rigorous control of test conditions by careful
  choice of subject populations, adequate replication,
  and adherence to experimental design is essential
  in toxicology testing.

In spite of such shortcomings, it would be inadvisable to abandon laboratory toxicity testing. However, it is important to utilize such data with full cognizance of their limitations. While standards calculated in such a manner lack the logical basis to serve as appropriate standards for the protection of the environment, they can serve as first approximations, subject to verification by monitoring and verification in those ecosystems to which they are to be applied.

Since the national criteria do not consider interactions between species and with the environment,

- site-specific standards which are direct
  conversions of national criteria can only be
  considered as provisional standards, subject
  to field verification through monitoring.

To transfer bioassay results to field conditions, it is impor- tant to consider that the response of a system to a fixed set of conditions may change because of an offset in time. In general, this is a difficult property to establish, but it cannot be simply assumed. Procedures to establish this should be incorporated into protocols.

- Minimally, only data obtained at comparable
  times of day, year, etc. should be compared.

Issues Related To Toxicity-Based Data

Test Species Selection

A critical component of the Guidelines for Deriving Site-Specific Water Quality Criteria is the development of a minimum data base on the acute toxicity of chemical to species selected from eight taxonomic families. These families in freshwater include four fish families, a planktonic crustacean, a benthic crustacean, and a representative of a phylum other than Arthropoda or Chordata. To develop a final salt water acute value, species must also be tested from eight families so that all of the following are included: two families in the phylum Chordata, a family in a phylum other than Arthropoda or Chordata, either the mysidae or penaide family, three other families not in the phylum Chordata or any other family. The apparent objective of requiring acceptable test results for species in eight different families to make up a minimum data set seems to be based on a desire to have results from a cross section of aquatic life (i.e., several families from different phyla). This is apparently to recognize the fact that aquatic organisms differ in their responses to chemicals or that what is needed is a definition of the range of responses. Since it is not possible to test all aquatic life to derive acute toxicity information, the Guideline establishes eight species as surrogates for all untested species. Can test results from eight species adequately define the range of response of all species? Recently an EPA-OTS sponsored workshop on the surrogate species concept concluded that individual species are not necessarily representative of any larger subset of different species; however, a group or cluster of species may have a relationship to the probability of effect in a wide range of other species. The surrogate species cluster concept was endorsed by the workshop participants and appears to be in concert with the approach used in the Guideline. The surrogate species concept was developed by taking an introspective examination of aquatic toxicology data bases for a variety of chemicals with a variety of organisms. Don Mount and Wes Birge examined these data bases and concluded that acute toxicity test results on 4-5 species, irrespective of taxonomic classification, adequately defines the range of response likely to be found, even if a larger number of species were tested. Therefore, it appears that requiring as a minimum data set toxicity data on species from eight (8) families has some basis and is partially defensible.

However, there are several potential problems with the concept of using the taxonomic level of family as a means of selecting species to test. For example, individual species within a

family may possess quite different toxic responses. The Guideline, therefore, leaves too much flexibility in the choice of species to test. With the existing guideline it would be possible for an unscrupulous person to select an insensitive species in each family and thus derive a final acute value which is higher than would have been determined if more sensitive species had been selected. Perhaps the guideline should require that if information is known about the range of sensitivities of species in a family to a candidate chemical or to a chemical of similar structure that those species exhibiting a sensitive response should be used. Likewise, the Guideline should recommend that keystone species and recreationally and/or economically important species be included in the minimum data base for determining the final acute values. Keystone species are species whose elimination from a particular aquatic ecosystem would cause a drastic collapse of the system. A final criterion and perhaps the most important criterion in selecting species to determine the range of response to a chemical is that the species selection should be based on physiological and/or biochemical characteristics. Species making up the minimum data base should possess a diversity of physiology and/or biochemical profiles. Simply requiring testing of representatives from eight families of aquatic organisms does not necessarily insure a wide diversity of physiological and biochemical types.

The Committee is concerned that the Agency fails to communicate a clear understanding of the difference between selected species and geographic specificity.

In addition to questions related to the geographic distribution of species and their varied responses to toxic substances, unique ecosystem characteristics also vary geographically. Clinal variations in terms of latitude and elevation contribute to the potential for ecosystems to absorb, transport, degrade and sequester toxic substances. Factors related to stability, diversity, resilience and the amplitude of disturbance from the system trajectory can be geographically specific and in the context of this report, site-specific. The impact of altering national water quality standards based on site-specific testing may be time related as a result of geographic specificity.

Recommendations for the collection of site-specific infor-mation to vary national water quality criteria should consider geographic specificity. These differences are related to the many physical and biotic components which influence ecosystem function and consequently how systems might be more or less

responsive to altered pollutant input without being degraded
from their current or potentially higher human use.

Statistical Issues in the Site-Specific Water Quality Criteria

For a given pollutant, the guidelines for deriving site-
specific criteria are adaptations of the corresponding
national guidelines. While it is not appropriate to review
the national guidelines here, it is necessary to briefly
state their statistical basis to put the site-specific
methodology in context.

The national guidelines define a two-part criterion: one
part is intended to be protective against acute effects; the
other is intended to be protective against chronic effects.

The acute value is found by way of a distributional approach,
taking the maximum instantaneous concentration as one-half
the estimated fifth percentile of the distribution of family
geometric mean LC_{50} values. The chronic value is often
used as a maximum 30-day average concentration. It is calcu-
lated by dividing the estimated fifth percentile of the
distribution discussed by an estimated acute-chronic ratio.
For compounds that bioaccumulate, the 30-day average concen-
tration may be determined as a concentration to protect
organisms which feed on aquatic life, if this is lower than
the chronic number.

The national guidelines contain provisions for adjusting the
criteria as a function of variables, such as hardness, that
affect toxicity. They also provide a range of sensitivities
by insuring that the family mean LC_{50} values satisfy a
minimum data base requirement.

Essentially every feature of these calculations has been
subject to some reservations. The principal comments are the
following:

1. There is no clear statement of the degree of
    protection the resulting criteria afford, beyond
    the claim that most species are protected most of
    the time.

2. The species on which the LC_{50}'s are available are
    not representative of the nation or of any
    particular site.

3. The method for calculation of the fifth percentile is unduly sensitive to the number of points in the distribution of average LC_{50}'s.

4. The average acute-chronic ratios are not statistically or biologicaly valid.

5. The reduction of the fifth percentile by one-half is not justified by any scientific analysis.

6. A 30-day average concentration does not protect against effects which can occur over a shorter time.

Related criticisms concern the criteria for admitting an LC_{50} to the data base on which the criteria are calculated, the choice of family versus species means, and alternative choices for estimation of the fifth percentile.

All of these comments are relevant to the site-specific statistical methodology. However, the statistical procedures for the site-specific criteria cannot be substantially improved in the absence of local data by which relevant ecological dynamic states may be defined, measured, and modeled. For this reason, we do not recommend that the basic calculation procedure be modified. Nevertheless, the validity of the comments must be accepted, and it must be concluded that the site-specific criteria will not, as a matter of logical necessity, protect the integrity of the biological community.

Desired Environmental Quality--Definition of Environmental Integrity

The Clean Water Act mandates the protection of the physical, chemical, and biological integrity of the environment. These general goals have not been sufficiently translated into specific guidelines based on the required degree of environmental protection commensurate with the intent of the act. It is important that these goals be stated in terms of environmental parameters rather than laboratory based parameters, so that the performances of various protective strategies can be evaluated for their efficacy.

The development of a more detailed statement of goals for the protection of environmental integrity is not easy, and because of limitations in time, the Committee was not able to develop

a set of operational goals which could be readily adopted by the Agency. However, the Committee was able to identify a number of important issues.

Absolute protection of the aquatic environment from human impact may be considered to be our intangible goal. However, such a goal is clearly unattainable, even if there were no economic, societal, or political constraints. Any species, humans included, cannot exist without affecting the environment or being affected by it. The need for strategies to protect the environment is strongly influenced by human population densities and by the effluvia of municipal and industrial activities which are associated with human life-styles. While it is not possible to return to an environmental state which is based on the premise that humans have never existed and do not produce a present impact, it is possible to reduce human impacts.

The development of operational guidelines for the protection of the integrity of the environment needs to consider a number of factors. Ecosystems are dynamic entities, and thus they exhibit natural variability over time and space. When an ecosystem is challenged by pollutants, it does not necessarily stop functioning, but enters a new dynamic state and, in doing so, may maintain its integrity. To date, the new dynamic state has been judged not only by objective parameters, but has also included considerations of the subjective desirability of the new ecosystem state, based on its suitability for single species, such as trout.

Therefore, the normal variability of ecosystems and our generalized expectations of the required degree of protection, coupled with the intent of using the ecosystem for a number of purposes, results in a situation filled with conflict when one attempts to set guidelines for the protection of the environmental integrity. To prevent greatly differing interpretations of the required degree of protection by every state, municipality, and industry, the Agency should establish further guidance based on environmental parameters.

Specification of the Environmental Protection Problem

Rather than developing site-specific standards from laboratory toxicity data alone, dynamic perspective is needed to improve EPA's present toxicology-based approach. An environment, to be protective for a certain set of uses, consists of sets

of ecological conditions, species, resources (including
toxicants of interest), and interrelations within and
between these categories--all dynamically changing with time.
Thus, environments are dynamic systems driven by sets of
inputs to generate time dependent behavior, which can be
viewed as outputs. Not all outputs can be measured to assess
the state of health of an environmental system; the problem
is to assess the condition of the environment from a restricted
set of carefully chosen diagnostic variables. Protection or
nonprotection of these variables must reflect success or
failure, respectively, in protecting the subject environment.
To achieve this, the diagnostic variables must be chosen so
that this protection bears both sufficient and necessary
relationships to the larger set of ecosystem variables which
denote an environmental condition.*

Environmental changes occur because of the influence of both
conventional pollutants and toxic substances. These changes,
constituting perturbation dynamics, can be compared to nominal
(unperturbed) behavior to provide a measure of impact. When
deviations exceed prescribed bounds, represented by standards,
the subject environment is considered unprotected; otherwise
it is protected up to the specified standards. The objective
of environmental protection, then, is to maintain differences
between perturbed and nominal behavior within limits defined
by the standards. Since it is inefficient and probably
impossible to measure all of the output variables of an eco-
system at the present time, the operational problem is to find
a subset of diagnostic environmental variables which reflect
the behavior of larger set of ecosystem output variables.
If the diagnostic output variables have been properly selected,
then as long as the diagnostic output variables remain within
specified limits (standards), the output variables of the
entire ecosystem would also remain within specified conditions.

Choice of Diagnostic Variables

The logical basis of choosing a set of diagnostic variables is
critical and should be approached through careful study of
each site-specific problem. Diagnostic variables should

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* A more extensive treatment in terms of system theory
is attached as Appendix 1.

- 13 -

embrace ecological and toxicological considerations, include system level abiotic and biotic variables, meet both short-term (acute) and long-term (chronic) diagnostic needs, include invariant elements to standardize across all possible national sites, and include variant elements which reflect site specificity.

EPA should pay close attention to conditional logic in establishing necessary and sufficient conditions required to relate protection of diagnostic variables to that of the larger set of ecosystem variables. The protocols developed should contain procedures which unequivocally establish these relationships for each site-specific problem.

Present practice has not generally included these important logical considerations, and

● there is a definite research need to better identify the appropriate diagnostic variables for the evaluation of environmental integrity.

Environmental Monitoring

Since it has been established that laboratory-based studies alone provide only an imperfect foundation for site-specific water standards, no matter what system of calculation is employed, environmental integrity needs to be assessed by means which are more closely connected to the environment. Therefore,

● the Committee strongly recommends a greatly expanded role for environmental monitoring in the development of site-specific standards.

There are three types of monitoring--chemical, physical, biological--and at least five purposes of monitoring. Baseline monitoring requires choice of diagnostic variables, determination that these are necessary and sufficient to represent ecosystem behavior, spatial and temporal design of sampling, and data assembly, analysis and presentation. Monitoring for impact detection is a different problem for chronic and acute effects, and, in addition to the above considerations, concerns definition of suitable standards. Compliance monitoring is an ongoing version of impact detection, but otherwise may be similar to impact detection. Monitoring to establish causality is a technically difficult problem and hinges on the logic underlying the monitoring effort. Specifically, a necessary

and sufficient relationship must be established between suspected cause and observed effect. Then, if the cause is present, the effect will be seen in monitoring data (sufficient), and if the cause is absent, the effect will not be observed (necessary). <u>Monitoring for prediction</u> is also technically difficult because it involves a time offset.

Protocols for Environmental Monitoring

Since the onset of monitoring as a vehicle for overseeing environmental integrity, the agencies involved (usually government at all levels) have moved slowly in making necessary changes in accepted protocols and methods. This is understandable in the sense that

1. baseline data are either wanting, scarce, or difficult to obtain;

2. governments have not, in general, carried out short-term or long-term research for meeting such methodology problems;

3. general acceptance by the scientific community is usually awaited prior to the institution of tactical revisions;

4. new chemical products and new toxicological problems are constantly emerging; and

5. ecology has only recently started to emerge from a descriptive to a systematic science--and is still emerging.

The results of this sequence of developments, along with an ever-increasing list of environmental knowledge requirements, have been a mixture of monitoring systems in dire need of technical and logical revision. Among the practices commonly in use are the following:

1. The monitoring of <u>all</u> possible factors and then using derived standards from best available information. This would include, for example, continued periodic measurements of metals, nutrients, plankton, bottom fauna diversity, whether or not they were pertinent, and then using fixed values, excursions beyond which constitute a violation.

2. The use of certain tools or criteria as monitoring bases which were once derived for a specific reason but which are now misused, in disuse, not up-to-date, or have been surpassed. There are many examples of these. An outstanding one is the setting of a single dissolved oxygen standard for all waters and the labeling of this standard as a "natural" level below which are levels of violation. An example of this technique is the current methodology of collection and measurement of oil and grease in native waters. Another is the inflexible adherence of some agencies to bottom sampling methodology and statistics despite bottom-type differences. This same attitude of inflexibility pertains to the use of a fixed array of bioassay organisms despite the varying conditions among areas under observation.

3. The use of a strictly toxicological approach to monitoring based only on LC_{50}'s or chronic health problems in specified organisms.

4. Monitoring for indefinite periods, which may be overly frequent or infrequent, and are not necessarily designed to include events of <u>natural</u> frequency.

The above approaches have within them strategies and tactics which need to be retained or updated. Rejecting them out-of-hand without review is not constructive. (The value of this, if nothing else, will be to point out complex problems and approaches to be avoided. This is the principal "value" of the several interim site-specific reports which we have looked at.) At the same time, Agency use and accompanying enforcement, when done in the narrow sense, is not only unconstructive but would also diminish progress towards our understanding of environmental reality. What is strongly suggested here is an intensive, periodic review and update with accompanying rules and technological changes, as needed, of monitoring protocols. (This is an essential step after an agreement is reached between a State and EPA.)

Periodically, there must be a review followed by necessary revisions and updates. It is also essential that the review group be composed mostly of people not employed by the government. These remarks apply to all governmental agencies that monitor, which have a national overview, and whose regional offices often are looked upon as ultimate arbiters in regional ecology.

- **What is needed first is a full description of the reasons for the existence of monitoring programs at strategic and tactical levels.**

Second, the protocols and technologies adopted must justify these reasons, be altered until they do so, or be rejected. By way of example, the phrase "best available information--or method" should only be a signal of needed information and a direction indicator for finding it. Also, the laws promulgated on clean water, air, etc. should be anticipated enough so that the kind of monitoring protocols needed as part of their body should also be anticipated. A sincere effort in this direction is not only economical but also avoids methods that are on the one hand expedient, but on the other do not fulfill the principles of whatever philosophy is derived. The case studies examined are an example of what can happen when time constraints are put on an investigatory team. Anticipation of method application, seasonal and diurnal changes, experimental planning of sampling techniques, and tryouts in the field should be an integral part of every case study.

Issues

1. The Purpose of Monitoring

Unless there is an end-product that is usable and justifies the principles directly or indirectly, then we only have monitoring for its own sake or to satisfy an arbitrary standard. In addition to practicability, monitoring should provide data and information for storage and retrieval in a cross-referential sense. Let us consider what it is that monitoring is supposed to show.

- a. **Monitoring should indicate when a designated level of substance is reached in a medium or organism, or, where pertinent, in a system of media and/or organisms.**

With respect to organisms, the protocol should involve one or more of the following types:

    1) organisms which concentrate a substance(s) significantly and are a significant part of the food chain;

    2) those which show pathology and which are a significant part of the local system. (This may eliminate transient forms and those which occur infrequently and in insignificant numbers.)

- 17 -

With respect to media or systems, the following factors should be considered:

1) Is the medium itself part of a system, i.e., in this case if it is part of a stream, are particulates in a transient state or prolonged suspension state? Is the bottom scoured and/or penetrated by upper waters and to what extent? Are the sediments chemically active, etc?

2) Is (are) the substance(s) dissolved? In suspension? On or associated with particles? In the sediments and, if so, distributed in interstitial spaces, sorbed on surfaces, ionically bound, etc?

3) Is the substance distributed in more than one of these states and, if so, is there equilibrium, steady state, or some other reactive order of distribution, half-life, etc?

● b. <u>Monitoring should indicate the factors which modify the chemistry/physics and the biological effects significantly.</u>

This may involve the status of the local environment of the substance(s) monitored or the effect of the substance on the organism tested.

One must select among the various possible chemical and physical factors for those which involve quantitative correction of measurements, e.g., temperature of water and oxygen solubility, salinity and pH, etc., and those which bring about chemical changes, e.g., pH and ionization or solubility, dissolved oxygen content and nitrification, etc. It must also be remembered that these factors may affect the uptake processes of organisms or change the chemistry of a concentrated substance in a food-chain organism when the latter is consumed by a grazer or predator.

Usually after a certain amount of background study, many of these factors can be spot-checked and the number of analyses reduced, or, because of interactions, control factors may be predictive of other ones, e.g., the components of an alkalinity system, once the interaction is established, can be calculated after pH is measured.

Here one must not forget the importance of seasonal change as an overall control of environmental inter-action.

There are enough ecological studies available that one can select influential physico-chemical factors. Many agencies do not do this and end up with quantities of multiple factor data for which there are no realistic correlations. The great disadvantage of this is not so much the accumulation of useless information as the fact that, while carrying out this type of monitoring function, influential factors may be downplayed or overlooked.

    c.   Monitoring should show which bioeffects are indicative of the <u>degree</u> of severity of the toxic problem.

Up until now almost all reports deal with LC_{50}'s and chronic effects. What may be being overlooked is the body of information that deals with other effects:

    1)   Behavioral changes of movement, habits, feeding, etc.

    2)   Degree of growth using possibly mark-and-recapture techniques among animals; mark and measurement among plants; mass increase in microplant or microanimal populations over short periods; periodic size distribution measurements among captured or sessile populations.

    3)   Fertility measurements which can be done in various ways among a variety of organisms. The question arises as to what adverse effects on fertility can show. First, among small organisms, they are the early warning systems that can predict the onset of lethal and chronic effects among larger organisms. Second, they may be measureable and take place at known fractions of lethal and chronic effects when occurring either in micro- or macro-organisms. Third, they may be indicating that lethal and chronic events are being partially masked in some organisms by an environmental effect, e.g., weak sorption on clay particles. Finally, these effects,

though not lethal at the moment of measure-
ment, may either result in the death of the
organisms individually or portend the limit-
ing or elimination of a population affected.

Judicious use of a. through c. should yield a series
of criteria that will be predictive when used together
with proper protocols derived from the other issues
to be discussed. The effectiveness of this use of the
criteria can be evaluated by field measurements of
selected species' population samplings. Some of the
measurements implied are done in the field (mark and
recapture); some are done in the laboratory (behavior
studies). In the latter case, however, if one is
fortunate in the choice of organisms, the laboratory
study indicates the type of behavior expected in the
presence of known levels of toxic agent.

2. Organisms to Use in a Monitoring Study

The original idea of the choice of test organisms
for toxicity studies was to give an array of reactions
through the phyla of animals that would cover important
vertebrates and invertebrates and provide a usable data
base. These then, on the bases of LC_{50}'s and chronic
effects, could be applied to the setting of standards
for levels of toxic substances in the environment.

With time, there has been a tendency toward standardized
use of certain species which apparently have been chosen
because of what has been regarded as their "importance"
and/or their sensitivity. The implication here is not
that these were or are bad criteria or that their use
should be disregarded. What is implied is, for site-
specific protocols, that regional authorities should be
leaning more toward local ecology both abiotic and
biotic. This means that one should learn which
population systems are operative, what is the impact
on species diversity and biomass, which trophic levels
are being dealt with, and, finally, rather than
determining the sensitivity of the organisms to a
particular toxic compound, first determine how important
the various organisms are in the system and then choose
among the most important for the desirable degree of
sensitivity.

The term "importance" can be given statistical or
numerical value just as can "sensitivity." Usually
it has to do with the largest number of particular
species, but this can also be weighted in terms of
how they function in the different trophic levels.

Thus, for a particular water/wetland system, one
should determine a differential quantification of
the operative systems, including, among the flora,
at least the microplants. It is understood, of
course, that some organisms are more motile and
transient and require number estimation techniques
that are more consistent than accurate. After
the quantification study has given sufficient data
to give a usable view of the system, an importance
figure should be ascribed both to those species
present in large numbers and to those important in
the trophic levels. Then a system should be devised
for choosing among them with regard to sensitivity,
taxonomic level, and species of particular interest,
probably in that order.

3. Frequency of Monitoring

As mentioned above, monitoring agencies may frequently
employ a variety of measurements, especially abiotic
ones, and end up with counts for which there is often
no correlation or which do not give a truly good
background picture. This criticism also applies to
the frequency with which the measurements may be
taken.

The following apply to physico-chemical measurements
of wetlands, water bodies, etc.

    a. Time of day. Such factors as dissolved oxygen
       and pH change diurnally, and at any particular
       site there should be night as well as day
       measurements. It is not necessary, however,
       to measure these on an hourly basis, unless
       there is a known dependent reaction occurring,
       e.g., elevated pH causing precipitation of
       carbonates. Otherwise, this is a waste of
       effort. Therefore, one will plan the necessary
       hours for abiotic determinations initially
       and cut them down to the minimal number of
       subsequent determinations.

b. Season. It may also not be necessary to
    measure conditions on a monthly basis
    continually. For the most part, measurements
    at the peak of the season and at the
    transition should tell what water condition
    changes will be. After these are established,
    it may only be necessary to measure the peaks.
    One should also be on hand to measure major
    occurrences such as major storms, spring rains,
    snow melts, etc.

c. If gradients from effluents to spring rain
    runoffs occur, these should also be noted and
    monitored.

d. If there is an intimation of possible stream
    contamination of a known type from a known
    source, the Agency should be prepared on short
    notice to measure downstream gradients.

e. It is apparent from 19 site-specific studies
    submitted to us for review that, to assure
    the information necessary to retain environ-
    mental integrity, some tighter specifications
    for the conduct of such studies should be
    written. Due to seasonal, diurnal, and
    meteorological or hydrological events,
    chemical and physical parameters are
    constantly changing, as are the populations
    of organisms in a stream. It seems logical
    that either sampling and monitoring must
    bracket these variations or other studies
    must be referenced to deduce how these
    changes with time will alter response of the
    organisms to toxic substances or alter the
    physical or chemical state and availability
    of those substances to affected organisms.

f. Also to be taken into consideration must be
    the physical dimensional effect on seasonal/
    time measurements, i.e., the depth of water,
    the influence of bottom in shallow waters vs.
    deeper waters, speed of currents, influence
    of shoreline, vegetation overhanging shore,
    etc.

- 22 -

With respect to organisms, measurements should be made according to the above criteria if _advisable_ only and at other such times as may elucidate some part of the life history that may be pertinent to the toxic study of concern.

4. Monitoring Feedbacks

One should be able by means of control situations and localities to set up a background picture or noise level of the important parts of the system under study. One must also decide on the levels of significance which depart from this "norm," both in degree and in time. From this, one should be able to decide when some pathology is occurring at the site area and possibly which groups of organisms are most affected. Selected tests for toxicology, as discussed above, may then ensue.

SPECIFIC PROCEDURES IN THE GUIDELINES

The proposed guidelines suggested that site specificity could be achieved through the application of one or more procedures, which can be briefly summarized as:

1. A "Recalculation Procedure," which recalculates the criteria by taking account of differences between the species tested as part of the national data base and those species which actually occurred or were expected to occur at a local site. This procedure is basically an extension of the methods for the derivation of the national criteria based on laboratory toxicity testing.

2. An "Indicator Species Procedure," in which acute bioassays are conducted in site water and in defined laboratory water to derive a ratio which represents the impact of local water quality. This ratio is then applied to the national data base.

3. A "Resident Species Procedure," in which the national toxicity data base is repeated for both acute and chronic toxicity using local species in site water.

4. A "Heavy Metals Speciation Procedure," which depends upon the metal concentration in a $0.45\mu m$ filtrate, rather than total metal concentration.

5. A "Historical Procedure," which sets the site-specific standards at local concentrations when it can be determined that these concentrations produce no adverse effects. This procedure is still under development.

6. A "Final Residue Value Procedure," which is essentially identical to that in the national water quality criteria methodology and is based on laboratory data.

These methodologies were examined in detail, and their evaluation follows.

Recalculation Procedure

The Recalculation Procedure is intended to account for the fact that some species which are represented in the national data base may not be present at a site. To remedy this, the site-specific criterion is calculated by removing the average LC_{50} values of the absent species from the data base on which family mean LC_{50} values are calculated. If removing these species reduces the data base below the national minimum data requirements, new bioassays would be run, or the national family value retained.

The Water Quality Handbook presents several caveats concerning
the validity of any resulting criteria. The principal
statistical hazard is that, if the number of families is
reduced, the site-specific instantaneous maximum will be lower
than the national criterion, all other things being equal.
Since the rate at which the distribution increases necessarily
goes up as the number of data points goes down, the estimated
fifth percentile will necessarily be lower. Consequently, it
does not appear that this type of effect can be eliminated as
long as interpolation from the empirical distribution function
is used to define the instantaneous maximum.

Since the recalculation procedure is based solely on laboratory
toxicity data, and is, in fact, a modification of the national
water quality criteria guidelines, it is subject to the same
criticisms as the national guidelines. The major criticisms
are that interactions and ecosystem properties are ignored, and
that the taxonomic basis for the selection of test organisms
does not reflect ecological structure or importance.

Indicator Species Procedure (Water Effects Ratio)

This procedure assumes that differences in water chemistry at
individual sites may modify the availability and thus the
toxicity of specific chemicals.

The Indicator Species Procedure is intended to cover the case
in which the range of sensitivities of resident species is the
same as that in the national data base, but the toxic effect is
modified by the characteristics of site water. The procedure
allows three ways to calculate the chronic value which usually
determines the 30-day average concentration. If available, the
national acute-chronic ratio may be used. Alternatively, three
matched pairs of acute and chronic tests on at least one fish
and one invertebrate may be run using site water. The geometric
mean of these three acute-chronic ratios is then used as the
acute-chronic ratio for estimating the chronic level. Finally,
a State may conduct acute and chronic tests on a fish and an
invertebrate in both laboratory and site water. From these
data a geometric mean chronic water effect ratio is calculated,
then multiplied by the national acute-chronic ratio to find the
ratio applied to the estimated fifth percentile.

The acute value is found by adjusting the national value by the geometric mean of the ratios of the site water LC_{50} to laboratory water LC_{50}, if this ratio is significantly different from one. Otherwise the national value is the site-specific value. The site water and laboratory water LC_{50}'s, which are averaged, are based on one fish and one invertebrate species.

As described in the Water Quality Handbook, the choice of species to be tested and the subset of the national data base on which the criterion will be derived are determined by the State, depending on the situation at each site. There is general guidance on these choices through a schematic workplan, and caveats are given concerning the main features of selecting the indicator species and running the required laboratory tests.

Because of the freedom granted the States in designing the studies for implementing the Indicator Species Procedure, it is difficult to assess the validity of criteria developed from this protocol. Unless Agency scientific staff are involved in planning and conducting studies implementing this procedure, different criteria may result for similar sites, due to different choices among the allowable alternatives. It would be helpful to provide a decision-tree for this procedure, together with guidance on the conditions under which each branch would be followed.

Statistically, the Indicator Species Procedure raises the question of the appropriateness of the small sample sizes for species and the ratio form of adjustment for acute and chronic water effects. The assumption is that the average of two or three species, some of which may be in the same family, provide data sufficient to adjust the distribution of family means by a constant of proportionality. Whether this can be successfully done, and for which pollutants, is a basic research issue which has not been resolved at this time.

From a biological and toxicological point of view, criticisms of the Indicator Species Procedure include (1) the assumption, in order to use it, that no species response differences exist between resident and national data base species, which is unrealistic; (2) "acceptability" for nonresident indicator or surrogate test species is not defined; (3) apparently, any two species of a fish and an invertebrate may be used in testing for chronic toxicity; and (4) the methods presented in this section do not provide adequate procedures for assessing the relative impact of various site waters on bioavailability and/or toxicity.

- 26 -

At a more specific level, these procedures call for simultaneous acute tests in "site and laboratory dilution water," but neither of these waters is adequately defined. This is of particular concern for site water where the underlying assumption seems to be that waters collected from a site in an unspecified manner, and transported and stored under unspecified conditions, will bear some resemblance to the water on site. The chemical characteristics of site water, however, are in large part a consequence of other factors at the site (e.g., pH, DO, salinity, temperature), which determine the availability of the chemical in question (Sunda and Hansen, 1979) and may change considerably on storage.

Moreover, water on site is in equilibrium with the substrate of that site, and disruptions of these equilibria among transport and storage may alter the availability of existing chemical species in the "site water." These modifications and the absence of the original substrate make extrapolations from laboratory tests of site water to actual on-site conditions difficult.

Also many of the factors mentioned above vary in site waters on a daily basis and seasonal basis, which could cause substantial differences in water effects ratios depending upon the time of day or year in which the water was collected. (See, for instance, draft report of Iowa River.)

These issues are mentioned in passing in the procedure (see 3-34). This discussion, however, merely cautions that the site water should be used "as soon as possible after collection" and that "care should be taken to maintain the quality of the water" and any changes should be "measured and reported." While acknowledging the potential for problems, these statements provide little useful information as to appropriate storage time, transport conditions, or the type or degree of changes which are considered acceptable/unacceptable. Furthermore, this discussion implies that flow through testing on site is more appropriate for dealing with problems in diurnal water quality cycles but does not address the actual issues of diurnal cycles or how this procedure would be applied.

Another issue left unaddressed in this section is the problem of sample contamination during collection and storage. Recent studies in marine chemistry have demonstrated that standard collection, storage, and analytic procedures resulted in metal contamination which produced metal concentrations several orders of magnitude higher than are now routinely measured in water samples (Patterson and Settle, 1976). It is also becoming

apparent that elevated metal levels due to metal contamination may be sufficient to inhibit growth in phytoplankton (Sunda and Guillard, 1976; Anderson and Morel, 1978) and to affect phytoplankton metabolism to the degree that primary production measurements are significantly altered (Fitzwater et al., 1982). A recent study with crab larvae also has demonstrated perturbations in growth at cupric ion activities just beyond ambient levels in the estuary from which the crabs were obtained (Sanders et al., 1983). These studies point out the importance of using carefully controlled clean procedures when collecting samples for water chemistry and biological effects studies. These clean procedures have been described in literature (Patterson and Settle, 1976; Bruland et al., 1979; and Fitzwater et al., 1982) and should be reviewed, modified, if appropriate, and incorporated in section B before it is implemented.

The use of acute/chronic ratios in this procedure also presents potential problems. In the first example on page 3-26, the site specific final chronic value can be obtained by dividing the national acute/chronic ratio into the site specific final acute value. This procedure assumes that the acute/chronic ratio derived from studies with laboratory water has some absolute reality and can thus be applied directly to data based, at least in part, on site specific assays. Recent studies with both metal and organic contaminants, however, make it clear that the mechanisms for acute and chronic effects may be quite different, and, as a consequence, these responses are not directly linked. While some consistency can be obtained in replicate experiments, where procedures are limited to a single variable, the validity of applying a national ratio to a site specific acute value, which is based on different and less defined procedures, has yet to be established.

A final point is drawn from the draft report on criteria modification for Selser's Creek, LA, prepared by J.R.B. In this report, water effects ratios were determined for Cd and Pb using the pigmy sunfish (Elassoma zonatum) and the grass shrimp (Palaemonetes kadiakensus). The water effects ratios determined using grass shrimp were 2 and 40 x higher for Cd and Pb respectively than were those determined using the sunfish. The reasons for these differences are not clear, but they serve to point out the potential problems with the water effects ratio as it is currently presented.

As a result of all of the shortcomings delineated above,

- <u>The indicator-species methodology should not be used for the development of site-specific standards until the concerns expressed have been resolved through research.</u>

Initially:
    (a) Procedures for collection, storage and testing of site water should be evaluated and a rigorous and standardized procedure should be implemented.
    (b) The problems of diurnal and seasonal variations should be addressed in a more specific and rigorous fashion.
    (c) General and specific limitations or problems in the procedure should be more clearly defined and their implications discussed. Ultimately the principles of this procedure should be incorporated in a totally redesigned methodology which is soundly based on ecological considerations.

Resident Species Procedure

The Resident Species Procedure essentially duplicates the complete battery of laboratory toxicity tests required for the minimum data base for the national criteria guidelines, but does so in local site water. Subsequently, a site-specific standard is calculated. The procedure is subject to the same criticisms as the national criteria guidelines and the recalculation procedure. However, if the appropriate precautions on handling of local site water delineated in the previous section are observed, then this procedure should account for many of the influences of local water quality on toxicity.

Heavy Metal Speciation Procedure

The national standards for metals are expressed as total recoverable metal based on laboratory data on total recoverable or acid extractable metal concentrations. Metals exist in a variety of forms, each with specific toxicological potential.

There is substantial evidence indicating that the availability and toxicity of aqueous trace metals is determined by the free metal ion activity rather than the total concentration of ions in solution (Sunda and Guillard, 1976; Anderson & Morel, 1978; Sunda et al., 1978). In spite of the title, however, section D of the Draft Water Quality Standards Handbook does not actually address the question of metal speciation. Rather it imposes an arbitraty size limit (0.45μm) below which metals are considered dissolved and thus potentially available.

As pointed out by Stumm and Brauner (1975), the filtration approach is not an appropriate procedure for addressing the question of metal speciation. Metal speciation is controlled, in large part, by the chemical associations of metals with inorganic and organic metal binding ligands (Sunda and Hansen, 1979). A crucial question in metal speciation is the strength with which metals interact with the various ligands, which is represented by the stability constant of each metal ligand complex. These constants are, in turn, highly dependent upon environmental factors such as pH, salinity, and DO (Stumm and Brauner, 1975; Sunda and Hansen, 1979), but are not directly related to particle size. Thus, metals associated with low molecular weight ligands with high binding constants may be less available than metals associated with larger ligands with lower binding constants. A probable example of this is seen in the draft report on criteria modification for the south fork of the Crow River, MN, prepared by J.R.B. In these studies LC_{50} values for total Cu were significantly higher than for dissolved, and attempts to use the filtration procedure would be inappropriate. Clearly the filtration procedure described in section D could greatly overestimate or under-estimate the available metals in a water sample depending upon the types and number of ligands available in the sample and the physical and chemical characteristics of the water. The current procedure, then, is both inadequate and inappropriate for addressing the questions of metal speciation.

In spite of problems with the current procedure, an under-standing of metal speciation is clearly central to the deter-mination of biological availability and toxicity of metals. Until recently, technical limitations have made routine studies of metal speciation in site waters difficult, if not impossible (Stumm and Brauner, 1975). A number of current studies based on electrochemical techniques, however, show excellent potential (Nurnberg, 1980). Techniques such as these, which directly address the question of metal speciation, should be pursued vigorously by EPA and should provide the foundation for any metal speciation procedures.

The issue of the biological availability of toxicants is not limited to the chemical and physical forms of heavy metals as they are affected by local water quality, but also includes the physical forms of organic pollutants.

- <u>The Agency should determine the most appropriate methods for estimating the concentrations of biologically active forms of pollutants for each individual pollutant.</u>

Historical Procedure

The historical procedure is still under development. Its intent is to set the site-specific standard at local concentrations, when it can be demonstrated that the local concentrations do not exert an adverse effect on the environment. The data required to support such a decision are closely linked to the data required for monitoring the safety of a provisional standard.

- **The Agency should complete the methodology for the historical procedure, including procedures for required monitoring, prior to setting the standard at in-stream concentrations.**

Final Residue Values

Residues of priority pollutants which are able to bioconcentrate or bioaccumulate in aquatic biota are of concern when the concentration of the residues reaches levels which exceed the guidelines for human consumption, or when the concentration of these residues reaches levels which may be hazardous to wildlife which feeds on aquatic organisms.

Action levels or guidelines for residues in fish and shellfish for human consumption are set by other agencies, but commonly are intended to protect most consumers with an ample margin of safety. These limitations also take into account that humans do not exist totally on food derived from aquatic organisms.

The requirements for the protection of wildlife which consumes aquatic organisms are less well established. In most instances it is assumed that the residue limitations which are adequate for the protection of human health are also adequate for the protection of other consumers of aquatic life. For wildlife species which consume aquatic life almost exclusively, the adequacy of such a level of protection may be questionable.

It is necessary to exert a degree of control over the residue concentrations in aquatic organisms to protect the consumers of aquatic life. Once criteria for maximum allowable concentrations in aquatic life have been established, the easiest and most direct procedure may be to monitor the concentrations in aquatic life to assure that these maximum allowable concentrations are not transgressed. Even though the concentrations of residues in biota are monitored regularly, to date the Agency has not incorporated the direct approach

- 31 -

into its criteria and standard-setting procedures. Instead, the Agency has chosen indirect approaches which seek to take advantage of the relationships between the pollutant concentration in water and the expected residue concentration in aquatic biota.

The estimation of the residue concentration in aquatic biota from water concentrations of the pollutant is based on the bioconcentration coefficient (BCF), which is the ratio of the residue concentration in the organism at equilibrium to the concentration of the pollutant in water. The bioconcentration coefficient can be estimated from the n-octanol/water partition coefficient (Veith et al., 1979), from laboratory exposures of fish or other aquatic organisms to the pollutant, or by field studies. The use of the partition coefficient for the estimation of the BCF is the least costly but is probably also the least accurate, in many cases. The use of laboratory exposures for the determination of the BCF under controlled conditions would seem initially to be the best way to determine the BCF; and, in fact, this procedure has become the mainstay in Agency approaches concerning the limitation of excessive residue concentrations in aquatic biota. The BCF values derived from field data have often been found to diverge significantly from laboratory-derived BCF values. To date, these discrepancies between laboratory data and field data have usually been cited to discredit the validity of the field data, when instead these discrepancies should lead one to equally question the validity of the laboratory data.

There are several important ways in which laboratory studies differ from field observations, which could drastically affect the determination of BCF values. The most important problem is that most laboratory BCF studies fail to recognize pharmacokinetic principles. Regardless of the fine details of the pharmacokinetic model which underlies the true uptake and clearance (depuration) of the pollutant, at equilibrium intake equals clearance. The excretion rate constant is, thus, one of the determining factors as to the duration required to reach equilibrium (Branson et al., 1975; Hartung, 1976; Neely, 1980). Thus, an exposure duration of approximately 4 excretion half-lives is required to approximate 95% of the equilibrium value. Most BCF studies are run for only 28 to 31 days, and equilibrium is assumed to have been reached. But, for a compound such as methylmercury, with an excretion half-life of about 1000 days, 12 years may be required before an equilibrium is approximated (Hartung, 1976). Because of the noise inherent in the analytical data, there is also a tendency to be overly hasty in pronouncing that equilibrium values have been achieved

in individual tests when pharmacokinetic data preclude such
a judgment. In addition, it is probable that the exposure
conditions in the laboratory vs. the field differ significantly.
Thus, the laboratory exposures ignore any impact of food webs
in reaching the residue equilibrium concentration. In
addition, in laboratory exposures the pollutant is usually
administered in a carrier, most commonly acetone, which may also
contain a surfactant to permit easier solubilization. Such a
method of administration may lead to an exposure consisting
of a combination of dissolved pollutant plus micellar pollutant.
In the field, the exposure would be to a combination of
dissolved pollutant, adsorbed pollutant on micro- and macro-
particulates, and pollutant incorporated in micro- and macro-
organisms. Clearly the exposure conditions are different,
and the current monitoring procedures in the laboratory and
in the field are not designed to evaluate the different routes
of exposure so that the effective dose can be calculated.

The heavy metals, excluding organo-metallic compounds,
generally do not bioconcentrate greatly in vertebrates. What-
ever bioconcentration does take place is not associated with
lipid solubility as the driving force, but most commonly
appears to be associated with facilitated transport mechanisms
which have evolved for required divalent cations, and which
do not appear to be absolutely specific for the required
metals. In the case of bioconcentration of metals by micro-
organisms, it is not always possible to determine whether the
metal is adsorbed onto the cell surface or whether it is
actually incorporated into the cell. Regardless of the exact
mechanism of bioconcentration of the heavy metals, the usual
laboratory test parameters tend to be simplisitic and usually
fail to account for the chemical and physical forms of the
heavy metal in the exposure water. The heavy metals are
usually added as a soluble salt in an acidified stock solution.
As soon as the stock solution is metered into the test aquaria,
the pH and the available ions for coplexation change. While
some of the metal usually remains in ionic form, the formation
of complex hydroxides, carbonates, and other insoluble salts,
plus the formation of chelates with organic trace materials,
can drastically alter the proportion of free heavy metal, and
therefore its availability. These conditions are probably not
representative of environmental conditions, where suspended
solids, different ions and different chelating agents may
predominate. In addition, in the environment most of the ionic,
salt, and chelate concentrations will have approached their
equilibrium state; while in the test situation, precipitates
and chelates are not likely to have reached that state. Again,
as for the organic compounds, the experimental state and the
environmental state are divergent.

Because of limitation in the experimental techniques discussed above, and because of the problems of duplicating food webs and other environmental phenomena in the laboratory, the field data should be examined more closely, and an effort should be mounted to make optimal use of all data. At first sight this might seem to be a very complex undertaking. However, all of the pharmacokinetic models, even relatively complex models which consider bioaccumulation via the diet (Pizza and O'Connor, 1983), reduce to linear models at or near equilibrium concentrations. Thus, as the equilibrium state is approached, the concentrations of xenobiotics found in aquatic organisms would become proportional to the environmental loading of that xenobiotic.

In the light of presently available information, more attention should be paid to the actual residue concentrations in the aquatic biota. These should ideally be considered as part of the criterion and standard-setting process.

- Field-derived bioconcentration coefficients, when coupled with an environmental fate model, can provide a more valid site specific method of calculating water quality standards.

- Final residue values need to incorporate site-specific physical and chemical characteristics which influence biomagnification.

Criteria for Site Definition

The Site-Specific Criteria Modification Process is designed to allow modification of national water quality criteria to local site-specific conditions. Review of the available documents and site-specific demonstration studies reveal that there is less to this process than meets the eye. In actual practice, definition of the site appears to consist of little more than selection of a sampling location from which to draw water for the site-specific bioassays within the impact zone of the pollutant in question.

A site must be specified on the basis of environmental parameters in conjunction with pollutant loadings. As such, it is unlikely to encompass whole States, but is more appropriately a stream segment or a portion of a body of water with relatively uniform characteristics. Boundaries (site modifications) may also include such things as changes in sediment or water characteristics (bottom of a large lake or bottom along a long stretch of river) which contain

- 34 -

organisms that are diurnal or spend part of the life cycle
in solubility sites (e.g., sediment binding of heavy
metals and some organics).

The amount of detail necessary to specify site conditions
depends, in part, upon which procedure for the modification
of the national criteria is selected. Because of the very
nature of a site-specific analysis, the description and
definition of the site will vary with the nature of the site.
The nature of the site definition will also depend very much
on whether the pollutant results from a point source or from
a non-point source.

Alternatively, the limits of a site could be defined by a
possible influence boundary and by a probable influence
boundary.

The possible influence boundary is the maximum area of
potential effect of aquatic life. The boundary should or
could be delineated by the detection limits of the known and
traceable contaminants from the discharge. The boundary is
here defined by a contaminant concentration to be calculated
or measured. The possible influence boundary constitutes the
farthest reaches of potential impact of the discharge.

On the other hand, the probable influence boundary can be the
minimum area of potential effect. The boundary could be
delineated by the concentration of contaminants known to be
detrimental to aquatic life.

The operational boundaries should be feasibly calculated based
on flow/volume/concentration information from discharge and
receiving waters. The concept of "site boundaries" clusters a
so called "extensible/contractable" template based on
contaminant concentrations. Areas within and distances between
boundaries are dependent upon receiving water and discharge
characteristics.

The probable influence boundary constitutes the limits of the
area to receive intensive study/monitoring/consideration
regarding protection of aquatic life and its uses.

The operational boundaries need not differ. This concept of
delineating two operational boundaries is intended as a frame-
work for priority and the intensity of testing of the aquatic
life and its uses. The probable influence boundary should
define an area which is to receive the most intensive
monitoring and evaluation.

Again these suggestions apply directly to idealized situations, i.e., one known discharge with known concentrations. However, the concept of boundary delineation based on contaminated concentration (calculated and/or measured) is in line with the NPDES Water Quality Guidance Criteria and flexes with discharge and receiving waters variations.

If site specific criteria are not to be used as enforceable numbers, the States may use them to develop effluent limits, water quality standards, etc. However, in the development of such limits or standards, a state is faced with an array of factors that has adversely affected earlier attempts to provide enforceable numbers.

A definition for a "site" states "it must contain acceptable quality dilution water upstream from the point of discharge if the site water will be required for testing" (EPA, 1983, p.5). The definition does not deal with sites that overlap nor with sites that extend beyond State boundaries. Acceptable quality dilution water upstream of a point discharge could be extremely difficult to obtain especially in water bodies that have numerous NPDES dischargers.

The General Work Plan presented in the Draft of the Water Quality Standards Handbook is much too simplistic and unrealistic in its approach to selecting sites. If the size of a stream segment is to make the examination of the site practical and that the stream segment should be affected by only one or two chemicals, then the general work plan would exclude such locations as the Hudson River and the Niagara River.

There are many other problems associated with the criteria to select sites. Biogeographic zones must be recognized, expecially as they might be influenced by toxic substances and/or multiple dischargers. Historical, physical, chemical, and biological data are not always available for the selected toxic pollutants. If background information is available, it must be carefully scrutinized for its reliability and accuracy. Previously utilized sampling techniques and analytical methodologies have not always provided the best data upon which to establish enforceable numbers.

The work plan, which is to initiate the criteria modification process at the State level, must have the components of a sound quality assurance program that would assure a more consistent implementation of the process throughout the country.

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APPENDICES

SYSTEM THEORY FORMULATION OF THE ENVIRONMENTAL PROTECTION
PROBLEM AND PROTOCOLS IN RELATION TO
SITE SPECIFIC WATER QUALITY CRITERIA

1. THE ENVIRONMENTAL PROTECTION PROBLEM

1.1 Introduction

The EPA's methodology for site specific modification of national standards for toxicants in aquatic ecosystems gives the impression of an almost heroic effort to deal with an impossible problem. The gap between the protocols and the need seems immense, but the problem is so poorly specified that this judgment is intuitive rather than being based on direct comparison of the protocols with a clear specification of the need. To evaluate the present methodology, as well as to assist progress toward more definitive ones, clearer definition of the fundamental problem is required. The purpose below is to attempt to formulate the problem in general system theory terms.

The objective of both national and site specific pollutant standards is to maintain the physical, chemical and biological "integrity" of the environment. This problem can be framed in terms of general dynamical ("state space") systems, defined by two expressions:

state transition function, ϕ: $Z \times X \to X$ or $x = \phi(z,x)$
response (output) function, ρ: $Z \times X \to Y$ or $y = \rho(z,x)$,

where $z(t) \varepsilon Z$, $x(t) \varepsilon X$ and $y(t) \varepsilon Y$ are inputs, states and outputs, respectively, and t is time. The transition function is usually approached through a differential equation, $dx/dt = f(z,x)$, whose solution is ϕ. The output function is an algebraic equation.

1.2 State Space Model of Environment

Define environment (ecosystem) E as a set of biological species S; abiotic substances R (resources, toxicants, etc.); interactions I, of three kinds: $S \times S$, $S \times R$; and $R \times R$; inputs Z, and outputs Y:

$$E = \{S, R, I, Z, Y\} .$$

A state space representation of E is

$$\phi: Z \times \{S,R,I\} \to \{S,R,I\}$$
$$\rho: Z \times \{S,R,I\} \to Y, \tag{1}$$

where, in general, some set X of state variables is to be formed from the elements of S, R and I. This expression models the environment E to be protected as a general dynamical system.

The general formulation (1) applies to a nonspecific, generalized environment. To establish site specific standards and protocols, specific systems $E = \{S,R,I,Z,Y\}$, must be considered, i.e., systems defined by specific state space formulations:

$$\phi: Z \times \{S,R,I\} \to \{S,R,I\}$$

$$\rho: Z \times \{S,R,I\} \to Y . \tag{2}$$

Here, $S \subset S$, $R \subset R$ and $I \subset I$. Any refined program for environmental protection with respect to toxic substances must take site specificity into account. Therefore, in principle, EPA's site specific program is in the right direction. The problem is converting principle to practice.

1.3 The Environmental Protection Problem

Let $p(E)$ represent the condition of a successfully protected specific environment, $E\varepsilon\mathcal{E}$. This can be defined in diverse ways according to a variety of criteria. Let $\bar{p}(E)$ be the complement, representing a damaged or degraded ecosystem according to the established standards. Similarly, let $p(Y)$ and $\bar{p}(Y)$ denote protected and unprotected system outputs, respectively, the means to assess the condition of E. The environmental protection problem then has several potential alternative formulations:

1. Find a subset of diagnostic system outputs, $\Omega \subset Y$, whose protection is <u>sufficient</u> to guarantee protection of E:

$$p(\Omega) \Rightarrow p(E) \qquad \text{(F1a)}$$
$$\bar{p}(E) \Rightarrow \bar{p}(\Omega) . \qquad \text{(F1b)}$$

The second form is the contrapositive of the first, and both are logical equivalents. In words, (F1a) states that protection of the chosen subset of output variables Ω is sufficient to guarantee protection of the site specific environment. For example, if the diagnostic output variables are several state variables such as a small set of species, $\Omega \subset S$, say the two species <u>Pimephales</u> <u>promelas</u> (fathead minnow) and <u>Ceriodaphnia</u> <u>reticulata</u> (a cladoceran), then (F1a) states that if the effect of some toxicant on these forms is acceptable by some specified standard, then this is sufficient to guarantee acceptable water quality with respect to this toxic substance in the particular water body E. Alternatively, (F1b) denotes that failure to protect E is sufficient to be reflected in nonprotection of the selected output variables. If the water quality of E falls below the specified standard, this will be seen as a toxic effect on the two diagnostic species.

At first glance, formulation (F1) seems an adequate basis for the development of site specific criteria. However, while it states the consequences of adequately protecting the output variables (F1a), and of failing to maintain acceptable water quality in the environment (F1b), it does not indicate the consequences of failing to protect the diagnostic species, $\bar{p}(\Omega)$, or of satisfactorily meeting the water quality standard, $p(E)$. As an example, prevention $p(\Omega)$ of mass fish mortality in a water body may be taken as sufficient to guarantee that a toxic substance has not, $p(E)$, reached acute concentrations. The attainment of such concentrations, $\bar{p}(E)$, would cause mass mortality, $\bar{p}(\Omega)$. However, a fish kill $\bar{p}(\Omega)$ can also be produced by unrelated causes, such as anaerobic conditions. Therefore, population mortality is an insufficient but necessary indicator of acute toxicant level. To compensate for this, the following formulation is needed.

2. Find a subset of outputs, $\Omega \subset Y$, whose protection is <u>necessary</u> to guarantee protection of E:

$$p(\Omega) \Leftarrow p(E) \qquad \text{(F2a)}$$
$$\bar{p}(\Omega) \Rightarrow \bar{p}(E) . \qquad \text{(F2b)}$$

(F2a) is the converse of (F1a) above, and (F2b) the inverse; again, (F2a) and (F2b) are logical equivalents. (F2a) states that in order to protect E, $p(E)$, it is necessary to protect the output variables, $p(\Omega)$. Maintenance of suitable water quality with respect to a toxicant is sufficient to guarantee noninjury up to the specified standard of the diagnostic species of the previous example, or the latter is necessary to guarantee the former. (F2b) states that failure to protect the output variables is sufficient to denote failure in protection of the ecosystem. If the diagnostic species register a

toxic response exceeding the defined standard, this means that water quality
has been correspondingly degraded.

Suppose fish mortality in absence of other causes (anoxia, disease,
etc.), $p(\Omega)$, is sufficient to signal an acute pollution load, $p(E)$. That is,
$p(\Omega) \Rightarrow p(E)$. Then it is clear that a low toxicant level, $p(E)$, is sufficient
to prevent fish mortality, $p(E) \Rightarrow p(\Omega)$. The fish populations together with
other relevant factors, $\Omega = \{$populations, disease, oxygenation,...$\}$, comprise an
output variable which can form the basis for a necessary condition to assess
environmental damage. That is, failure to protect Ω, $p(\Omega)$, denotes failure to
protect the ecosystem, $p(\Omega) \Rightarrow p(E)$, which is the logical equivalent of
$p(E) \Rightarrow p(\Omega)$. Whereas $p(\Omega)$ above was sufficient but not necessary to denote
environmental damage, $p(\Omega)$ here is necessary.

Clearly, a site specific protocol upon which to base an approach to
environmental protection should have the necessary, as well as the sufficient,
property. Therefore, the logical basis for environmental protection should be
as follows.

3. Find a subset of outputs, $\Omega \subset Y$, whose protection is both
<u>necessary</u> <u>and</u> <u>sufficient</u> to guarantee the protection of E:

$$p(\Omega) \Longleftrightarrow p(E) \qquad \text{(F3a)}$$
$$p(E) \Longleftrightarrow p(\Omega) \ . \qquad \text{(F3b)}$$

In the forward or reverse directions, (F3a) and (F3b) are again equivalent
pairs. The program to establish $pvp(\Omega)$ for a specific environment E comprises
a <u>site</u> <u>specific</u> <u>protocol</u>.

1.4 Models for Dynamical Systems

Reiterating from expression (1), any dynamical system with state vari-
ables X, inputs Z and outputs Y has the following representation of behavior:

$$\phi: Z \times X \rightarrow X$$
$$\rho: Z \times X \rightarrow Y \ . \qquad (3)$$

The state transition function ϕ is the solution of a differential or dif-
ference equation. Attention below will be restricted to the differential
(continuous time) case; the difference (discrete time) case follows a parallel
development.

There are, in general, two forms for $dx(t)/dt = \dot{x}(t) = f[z(t), x(t)]$,
$x \varepsilon X$, $y \varepsilon Y$, $z \varepsilon Z$ and $t \varepsilon T$, where T is a continuous time interval. One is <u>linear</u>,

$$\dot{x}(t) = A(t)x(t) + B(t)z(t), \ x(t_0) = x_0,$$

a special case where the solution is decomposable into an internal portion due
to state, and an external portion due to input:

$$x(t) = \phi[z(t), x(t)]$$
$$= e^{\int_{t_0}^{t} A(\xi)d\xi} x_0 = \int_{t_0}^{t} e^{\int_{\tau}^{t} A(\xi)d\xi} B(\tau)z(\tau)d\tau \ .$$

The second is <u>nonlinear</u>, the general case where state and input components of
behavior are inextricably compounded together:

$$\dot{x}(t)=A(t)x(t)+B(t)\zeta[z(t),\ x(t)],\ x(t_0) = x_0\ ,$$

with solution:

$$x(t) = \phi[z(t),\ x(t)]$$

$$=e^{\int_{t_0}^{t} A(\xi)d\xi}\ x_0 + \int_{t_0}^{t} e^{\int_{\tau}^{t}A(\xi)d\xi}\ B(\tau)\zeta[z(\tau),\ x(\tau)]d\tau\ .$$

The expression

$$e^{\int_{\tau}^{t}A(\xi)d\xi}\equiv\Phi(t)\Phi(\tau)^{-1}\equiv\Phi(t,\tau),\ t_0\leqq\tau, \tag{4}$$

describes change of state on the interval $[\tau,t]$, and constitutes a <u>state transition operator</u>. Behavior on $[t_0,t]$ can be modeled either by the <u>linear dynamical system</u>,

$$\dot{x}(t) = A(t)\ x(t) + B(t)\ z(t),\ x(t_0) = x_0$$
$$y(t) = C(t)\ x(t) + D(t)\ z(t)\ , \tag{5}$$

with transition function:

$$x(t) = \phi[z(t),\ x(t)]$$

$$= \Phi(t,t_0)\ x_0 + \int_{t_0}^{t}\Phi(t,\tau)\ B(\tau)z(\tau)d\tau\ , \tag{6a}$$

and response function

$$y(t) = \rho[z(t),\ x(t)]$$

$$= C(t)\Phi(t,t_0)x_0 + \int_{t_0}^{t} C(\tau)\Phi(t,\tau)B(\tau)z(\tau)d\tau + D(t)\ z(t)$$

$$= y_x(t) + y_z(t)\ ; \tag{6b}$$

or by the <u>nonlinear dynamical system</u>:

$$\dot{x}(t) = A(t)\ x(t) + B(t)\ \zeta[z(t),\ x(t)],\ x(t_0) = x_0$$

$$y(t) = C(t)\ x(t) + D(t)\ \zeta[z(t),\ x(t)]\ , \tag{7}$$

with transition function:

$$x(t) = \phi[z(t),\ x(t)]$$

$$= \Phi(t,t_0)\ x_0 + \int_{t_0}^{t}\Phi(t,\tau)\ B(\tau)\ \zeta[z(\tau),\ x(\tau)]\ d\tau\ , \tag{8a}$$

and response function

$$y(t) = \rho[z(t), x(t)]$$

$$= C(t)\,\Phi(t,t_0)x_0 + D(t)\zeta[z(t), x(t)]$$

$$+ \int_{t_0}^{t} \Phi(t,\tau)\,B(\tau)\zeta[z(\tau), x(\tau)]d\tau$$

$$= y_x(t) + y_{zx}(t). \tag{8b}$$

In these equations, x, y and z are vectors; x_0 is the initial state vector; A, B, C and D are matrices; and Φ is a state transition or fundamental matrix. Further theory regarding these formulations is available in Zadeh and Desoer (1963), Director and Rohrer (1972) and Mesarovic and Takahara (1975). Note that in the linear system, equation (6b), behavior is decomposable into zero input, y_x, and zero state, y_z, portions, but in the nonlinear system, equation (8b), such a separation is not possible. State and input are always compounded, $y_{zx}(t)$, in the function $\zeta(z, x)$. It is further observed that these models of dynamical systems, equations (5)-(8), are applicable to any system; specifically, to the entire ecosystem or environmental system, E, itself, or to any of its constituents, S or R, in equation (1).

1.5 Behavior Dimensions of the Environmental Protection Problem

Let E be the set of all natural or artificial (e.g., laboratory) environments, S the set of all species, and R the set of all resources, including toxicants. Let $y_{ijk}[t_0,t]$ be the nominal (unperturbed) behavior on interval $[t_0,t] \subset T$ of environment or environments $i=1,\ldots,e$, $i \subset v \varepsilon E$, containing set $j=1,\ldots,s$, $j \subset v \varepsilon S$, of the species S, and set $k=1,\ldots,r$, $k \subset v \varepsilon R$, of the materials R. Every specific ecosystem E, $E \varepsilon E$, has behavior dynamics defined by either equations (5)-(6) if it is linear, or (7)-(8) if nonlinear. The same is true for any set or element of its biotic or abiotic constituents, S and R, respectively, $S \subset S$ and $R \subset R$. Therefore, $y_{ijk}[t_0,t]$ denotes the behavior of any conceivable system; $y_{Eik}[t_0,t]$ represents dynamics of any entire site specific ecosystem, $E \varepsilon E$, and component behaviors are $y_{EJk}(t)$, $J \subset S$, and $y_{Eik}(t)$, $K \subset R$, as defined by either equations (6b) or (8b). With these notations, the behavioral dynamics of any system or any of its biotic or abiotic constituents, singly or in any combination, in any natural or artificial setting, and subject to any set of natural of artificial influences, may be formulated.

Let $z_{ijk}(t) \varepsilon Z$, $t \varepsilon T$, equation (1), be the nominal inputs to an environmental system or its constituents or constituent sets, and let $\hat{z}_{ijk}(t)$ represent perturbation inputs, such as when one or more toxic substances are introduced or increased. Then, the difference between nominal and perturbed behviors, $y_{ijk}(t)$ and $\hat{y}_{ijk}(t)$ respectively, measures the response of the system to its changed environment, $\hat{z}_{ijk}(t) - z_{ijk}(t)$. If the changed environment represents introduction or increase in the concentration of a toxic chemical, then $\hat{y}_{ijk}(t) - y_{ijk}(t)$ measures the toxic effect on the system or subsystem. Let σ_{ijk} be a vector (or scalar as appropriate) of standards for substance(s) $k \subset R$ in the presence of species $j \subset S$ in environment $i \varepsilon E$. National standards are denoted when $i=E$, σ_{Eik}, and site specific standards when $i = E \varepsilon E$, σ_{Eik}. The standards are constants, although to handle seasonal and other time dependent variability, they can in principle be time functions, $\sigma_{ijk}(t)$. They may also vary with different uses, $u \subset U$, where U is the set of all possible uses; thus $\sigma_{ijk}(t,u)$. In subsequent usage the standards will be considered as constants.

The environmental protection problem can now be formulated in dynamical system terms:

1. The _objective_ of environmental protection, maintaining the physical, chemical and biological integrity of the environment, is to minimize deviations (defined by the standards) between nominal, $y_{ijk}(t)$, and perturbed, $\hat{y}_{ijk}(t)$, behavior:

$$\min[\hat{y}_{ijk}(t)-y_{ijk}(t)] \quad \text{such that} \quad \|\hat{y}_{ijk}(t)-y_{ijk}(t)\| \leq \|\sigma_{ijk}\| , \qquad (9)$$

where $\|\cdot\|$ denotes vector norms, and the standards σ_{ijk} are expressed in terms of effects rather than concentrations to be in units compatible with those of the behavioral responses, $\hat{y}_{ijk}(t)$ and $y_{ijk}(t)$.

2. The _protocol problem_ in environmental protection is, following (F3) above, to find a small subset of output variables, $\Omega_{ijk}(t) \subset y_{ijk}(t)$, such that:

$$\| \hat{\Omega}_{ijk}(t)-\Omega_{ijk}(t)\| \leq \|\sigma_{ijk}\| \Longleftrightarrow \|\hat{y}_{ijk}(t)-y_{ijk}(t)\| \leq \|\sigma_{ijk}\| , \qquad (10a)$$

or alternatively,

$$\| \hat{y}_{ijk}(t)-y_{ijk}(t)\| > \|\sigma_{ijk}\| \Longleftrightarrow \|\hat{\Omega}_{ijk}(t)-\Omega_{ijk}(t)\| > \|\sigma_{ijk}\| . \qquad (10b)$$

These expressions, (9) and (10), then, comprise a theoretical formulation of the environmental protection problem.

2. ENVIRONMENTAL PROTECTION PROTOCOLS

2.1 Introduction

Note, in (10), that the protocol problem has three elements: (1) finding a suitable set of output variables, $\Omega_{ijk} \subset y_{ijk}$; (2) finding necessary and sufficient relationships between the variable sets, Ω_{ijk} and y_{ijk}; and (3) through the behavior of these variables, establishing the standards, σ_{ijk}, such that $\|\Omega_{ijk}(t)-\Omega_{ijk}(t)\| \leq \|\sigma_{ijk}\| \Longrightarrow \|\hat{y}_{ijk}(t)-y_{ijk}(t)\| \leq \|\sigma_{ijk}\|$. These are three distinct problems. The general protocol development approach of EPA gives great attention, however imperfectly, to (3), but virtually ignores (1), the extrapolation from a restricted set of observables, $\Omega_{ijk}(t)$, to the whole ecosystem, $y_{ijk}(t)$, and (2), establishing a logical rationale through which to relate the variable sets Ω_{ijk} and y_{ijk}. In the following sections an effort will be made to examine these problems connected with restricted diagnostic variable sets, and in addition general problems associated with laboratory based toxicology and monitoring. These discussions will provide a basis for then considering EPA's site specific procedures: recalculation, indicator species, resident species and heavy metal speciation.

2.2 The Diagnostic Output Variable Set

2.2.1 Choice of Variables

A fundamental problem, at the root of all further problems, is to find a restricted set of variables whose behavior under toxification will reflect, (F3) ecosystem condition. As the goal of maintaining physical, chemical and biological integrity involves three classes of variables, all three are potentially useable in the desired set Ω_{ijk}. EPA's approach focuses on only the biotic variables, S, and ignores (except for the specific toxic substances of interest) the resource set R.

In nature, the biotic variables are interrelated, $S \times S$; they also interact with the abiotic variables, $S \times R$, and the latter additionally

interact together, $R \times R$. The result of all these interactions, I, is integrated system behavior, $y_{ESR}(t)$. Any given ecosystem, $E \varepsilon \mathcal{E}$, *with a restricted list of species*, $S \subset \mathcal{S}$, and physical-chemical variables, $R \subset \mathcal{R}$, still has integrated behavior, $y_{ESR}(t)$. This integrated behavior, which may be linear, equation (6), or nonlinear, equation (7), involves measureable attributes at component and subsystem levels, $y_{Ejk}(t)$, $j \subset v \varepsilon S$, $k \subset v \varepsilon R$, as well as at the entire system level. Thus, both system level as well as component and subsystem level variables are available for selecting the diagnostic set Ω_{Ejk}.

EPA's approach to variable selection ignores ecological considerations, and focuses narrowly on nonfunctional, taxonomic ones. There are three basic criteria to consider in selecting a set of output variables:

Criterion 1. Resource variables, $k \subset v \varepsilon R$, which reflect ecosystem function (productivity, respiration, aerobic and anaerobic decomposition, etc.) should be included in Ω_{Ejk}. A focus on process outputs will yield variables which change on short time scales, giving an early warning capability.

Criterion 2. Biotic variables, $j \subset v \varepsilon S$, which are ecologically significant in ecosystem structure and function (keystone species, indicator species, commercially important species, procaryotes, etc.), should also be represented in Ω_{Ejk}. Again, emphasis on microbiota, which form the basis for production and decomposition processes, will contribute fast time variables necessary for early diagnosis.

Criterion 3. Taxonomic properties in the selection of biotic elements of Ω_{Ejk} should still be employed in an effort to bracket the range of toxic responses. There are two distinguishable elements of this approach, which emphasizes purely toxicological, as opposed to ecological, considerations: (1) The "laboratory white rat" concept. Some small set of organisms (species) or life stages should always be tested in all situations for comparative standardization. Pimephales and Ceriodaphnia are two that have been used for freshwater. Others are possible. The aim is a fixed data set under standard conditions for intersite comparisions. (2) The "surrogate species cluster concept". A somewhat larger set of organisms, spanning a wide range of physiological, biochemical and phylogenetic differences, should also be tested. These organisms for testing should be collected from field conditions most closely approximating those at the specific sites of interest.

EPA's approach of requiring representation from eight animal families from a selection of phyla is naive, although well motivated to obtain a wide range of toxic responses, because individual species within a family may possess quite different toxic responses. An industry bent on not cleaning up could meet virtually any standard by judicious choice of species, and still satisfy the taxonomic specifications. Also, specific requirements, such as that a salmonid fish or penaeid shrimp be represented, have valid ecological or economic rationales. If trout can tolerate a situation, the ecosystem which supports them cannot be too greatly perturbed; the trout is an indicator species. If shrimp are relatively unaffected, the same is true, and in addition commercial interests will not suffer; shrimp are an economic, as well as indicator, species. In selecting higher organisms, or those in which exposure to toxicants is indirect (e.g., as in food chain concentration), it should be remembered that response times would be expected to be correspondingly slowed.

Recommendation 1. EPA should carefully study the problem of choosing diagnostic vairable sets, Ω_{ijk}. These sets should:

1. Embrace ecological as well as toxicological considerations,
2. Include abiotic as well as biotic variables,
3. Meet both short term (acute) and long term (chronic) diagnostic needs,
4. Include invariant elements for standardization across all possible national sites, and
5. Include variant elements to achieve site specificity.

2.2.2 Variable Sufficiency and Necessity

The logical problems inherent in protocol development, i.e., (F1) vs. (F2) in Section 1.3, are difficult to solve in actual applications. The desired formulation is (F3), expressed operationally in expressions (10a) and (10b). But how are these relationships between the diagnositc variables, $\Omega_{ijk}(t)$, and the entire set of field variables, $y_{ijk}(t)$, to be established? Present EPA protocols ignore this altogether, assuming that whatever is true for $\Omega_{ijk}(t)$ also holds for $y_{ijk}(t)$. The fallacy here is readily apparent; for example, it was observed above that any desired response could be obtained by simply adjusting the member variables, e.g., one species instead of another in a given family. Both sufficiency and necessity must be considered.

Sufficiency. The two logically equivalent forms of this problem are, from (F1),

$$p[\Omega_{ijk}(t)] \Rightarrow p[y_{ijk}(t)] \qquad (11a)$$

$$\bar{p}[y_{ijk}(t)] \Rightarrow \bar{p}[\Omega_{ijk}(t)]. \qquad (11b)$$

The key to establishing (11a) is determining that (11b) holds. For example, suppose data obtained from observations made during $T<T'$ produced standards such that (11a) was demonstrated to be satisfied. That is,

$$\| \hat{\Omega}_{ijk}(T)-\Omega_{ijk}(T)\| \leq \| \sigma_{ijk}\| \Rightarrow \| \hat{y}_{ijk}(T) - y_{ijk}(T)\| \leq \| \sigma_{ijk}\| . \qquad (12a)$$

To make this demonstration it would be required, measuring all or most of the variables y_{ijk}, to establish (11b), i.e.,

$$\| \hat{y}_{ijk}(T)-y_{ijk}(T)\| > \| \sigma_{ijk}\| \Rightarrow \| \hat{\Omega}_{ijk}(T)-\Omega_{ijk}(T)\| > \| \sigma_{ijk}\| . \qquad (12b)$$

With (12b) demonstrated, the standards σ_{ijk} and the diagnostic variables Ω_{ijk} can be used during future times to infer, by monitoring $\Omega_{ijk}(T')$ only and not $y_{ijk}(T')$, that

$$\| \hat{\Omega}_{ijk}(T') - \Omega_{ijk}(T')\| \leq \| \sigma_{ijk}\| \Rightarrow \| \hat{y}_{ijk}(T') - y_{ijk}(T')\| \leq \| \sigma_{ijk}\| . \qquad (13)$$

Thus, $p[\Omega_{ijk}(T')]$ is sufficient for future protection of the ecosystem, $p[y_{ijk}(T')]$, per (11a). The initial effort to establish (12b) in some way must be made, however. In situ or laboratory microcosm experiments (e.g., in the mode of EXAMS, Lassiter et al. 1981) may be useful ways to approach this initial validation.

Necessity. From (F2),

$$p[y_{ijk}(t)] \Rightarrow p[\Omega_{ijk}(t)] \qquad (14a)$$

$$\bar{p}[\Omega_{ijk}(t)] \Rightarrow \bar{p}[y_{ijk}(t)]. \qquad (14b)$$

The latter, (14b), is the operative form for environmental protection purposes, but to establish it the equivalent form (14a) must be implemented. In the same manner as to achieve (12b) above, i.e. measuring all or most of the variables y_{ijk}, (14a) is demonstrated on $T < T'$:

$$\| \hat{y}_{ijk}(T) - y_{ijk}(T) \| \leq \| \sigma_{ijk} \| \Rightarrow \| \hat{\Omega}_{ijk}(T) - \Omega_{ijk}(T) \| \leq \| \sigma_{ijk} \|. \quad (15a)$$

That is, under conditions of whole system protection, the left hand side, no perturbations show up in the diagnostic variables, the right hand side. In effect, this would be the control of the field or microcosm experiment above to establish (12b), and achievement of (15a) also means that the form

$$\| \hat{\Omega}_{ijk}(T) - \Omega_{ijk}(T) \| > \| \sigma_{ijk} \| \Rightarrow \| \hat{y}_{ijk}(T) - y_{ijk}(T) \| > \| \sigma_{ijk} \| \quad (15b)$$

is valid, establishing (14b), the necessary condition. This can then be used in future monitoring to detect system perturbations based on perturbations in the diagnostic variables,

$$\| \hat{\Omega}_{ijk}(T') - \Omega_{ijk}(T') \| > \| \sigma_{ijk} \| \Rightarrow \| \hat{y}_{ijk}(T') - y_{ijk}(T') \| > \| \sigma_{ijk} \|, \quad (16)$$

allowing $\hat{p}[\Omega_{ijk}(T')]$ to be sufficient to establish future failure, $p[y_{ijk}(T')]$, to protect the ecosystem. This gives the required necessary condition, and (13) and (16) together comprise sufficient and necessary conditions, (F3), to protect an entire system based on the monitoring of a few variables.

    <u>Recommendation 2</u>. EPA should develop protocols and methods to establish (13) and (16) through field or microcosm experiments based on relationships, respectively, (12b) and (15a). Then, by monitoring the set, Ω_{ijk}, of selected diagnostic variables with established necessary and sufficient relations to the larger set, y_{ijk}, of system output variables, the status of ecosystems with respect to toxicant standards, σ_{ijk}, can be determined on a continuing basis.

2.2.3 Determination of Standards

    How should the values, σ_{ijk}, for different toxicants under different conditions be set? The notation for standards, σ_{ijk}, is efficient in allowing different combinations of conditions and toxic substances to be represented, as follows:

        1. Conditions. Heretofore, $i \varepsilon E$ has been used to index a specific ecosystem, $E \varepsilon \mathcal{E}$. Each specific ecosystem has a wide range of physical conditions in it over a period of time (temperatures, pH values, salinities, etc.). The index i can now be refined to refer to a specific set of more microscopic conditions defining macroscopic E. Thus, $i \subset v \varepsilon E$ refers to such a set of conditions rather than grossly to the whole ecosystem which has them as attributes.

        2. Biota. The index $j \subset v \varepsilon S$ refers to a particular combination of species; subsequent usage is unchanged from that previously established.

        3. Substances. Toxic materials may occur in polluted waters singly or in different combinations. The index $k \subset v \varepsilon R$ has previously denoted any possible combination of resource variables, including complex mixtures. This usage will be continued.

    Thus, σ_{ijk} refers to ecological conditions $i \subset v \varepsilon E$ in which biota $j \subset v \varepsilon S$ live utilizing, beneficially or harmfully, materials $k \subset v \varepsilon R$.

Suppose conditions differ, denoted, say, by i and i', i,i'$\subset$ vεE. Then $\sigma_{ijk} = \sigma_{i'jk}$ or $\sigma_{ijk} \neq \sigma_{i'jk}$, depending upon the effects of the changed conditions on the toxic responses of system output variables. Similarly, under different species combinations, j and j', where j,j'$\subset$ vεS, $\sigma_{ijk} = \sigma_{ij'k}$ or $\sigma_{ijk} \neq \sigma_{ij'k}$ are possible. Finally, under different resource settings, k and k', where k,k'$\subset$ vεR, $\sigma_{ijk} = \sigma_{ijk'}$ or $\sigma_{ijk} \neq \sigma_{ijk'}$ are possible. In principle, then, each of the standards σ_{ijk}, $\sigma_{i'jk}$, $\sigma_{ij'k}$, $\sigma_{i'j'k}$, $\sigma_{ijk'}$, $\sigma_{i'jk'}$ and $\sigma_{ij'k'}$ are all potentially different.

For illustration, with fixed i and j, suppose k denotes one set of substances, including toxicant A by itself, whereas k' represents the same resource set, except that now toxicant B is also present in addition to A. If A and B interact, synergistically or antagonistically, then clearly $\sigma_{ijk} \neq \sigma_{ijk'}$, and therefore

$$\| \hat{\Omega}_{ijk}(t) - \Omega_{ijk}(t)\| \leq \|\sigma_{ijk}\| \Longleftrightarrow \|\hat{y}_{ijk}(t) - y_{ijk}(t)\| \leq \|\sigma_{ijk}\| \quad (17a)$$

and

$$\| \hat{\Omega}_{ijk'}(t) - \Omega_{ijk'}(t)\| \leq \|\sigma_{ijk'}\| \Longleftrightarrow \|\hat{y}_{ijk'}(t) - y_{ijk'}(t)\| \leq \|\sigma_{ijk'}\| \quad (17b)$$

are two different things. σ_{ijk} and $\sigma_{ijk'}$ represent two different standards.

Therefore, since toxic respones may vary under different ecosystem conditions, i, species compositions, j, and resource combinations, k, standards should be set in contexts as close to those of the system of interest as possible. EPA has been partially responsive to this dictum by recognizing that different mixes of toxic materials may produce different toxicological results, and by correcting for experimental vs. site water differences through water effect ratios calculated in their site specific protocols.

Recommendation 3. EPA should develop site specific standards, σ_{ijk}, under conditions i, species compositions j, and mixes of substances k approximating as closely as possible the ranges of these factors met in the subject ecosystem. Protocols designed to compensate for variability met under field conditions, including possible temporal (e.g., seasonal) variability, should be carefully thought out as a basis for procedures that are realistic and practicable.

One of the major problems in developing such protocols is the transition from laboratory toxicology data to field situations.

2.3 Laboratory Based Toxicology

In Section 2.2.1 it was recommended that sets of diagnostic variables, Ω_{ijk}, include not only a few species selected for taxonomic spread, but in addition other variables based on ecological considerations. In general, however, toxicological tests are performed on selected species in specific life stages; therefore, they apply to only part of what should be a fuller set of diagnostic output variables. This part is subject to problems of extrapolating from experimental to field conditions.

2.3.1 Diagnostic Variable Dynamics

Just as the output variables, $y_{ijk} \varepsilon Y$, of a system are based on a corresponding set, $x_{ijk} \varepsilon X$, of state variables [equation (3)], so the restricted set of diagnostic variables, $\Omega_{ijk} \varepsilon Y$, has underlying it a corresponding restricted set of state variables, $x_{ijk} \varepsilon X$. If these state variables behave linearly on a time interval $T = [t_o, t]$, then equation (5) applies, yielding for the transition function of the restricted set:

$$X_{ijk}(t) = \phi_{ijk}(t)\phi_{ijk}^{-1}(t_0)x_{ijk}(t_0) = \int_{t_0}^{t} \phi_{ijk}(t)\phi_{ijk}^{-1}(\tau) B_{ijk}(\tau)z_{ijk}(\tau)d\tau , \quad (18a)$$

and for the response function:

$$\Omega_{ijk}(t) = C_{ijk}(t) \phi_{ijk}(t)\phi_{ijk}^{-1}(t_0)x_{ijk}(t_0) + D_{ijk}(t) z_{ijk}(t)$$
$$+ \int_{t_0}^{t} C_{ijk}(\tau)\phi_{ijk}(t)\phi_{ijk}^{-1}(\tau) B_{ijk}(\tau) z_{ijk}(\tau) d\tau . \quad (18b)$$

Equation (18a) corresponds to (6a), and (18b) to (6b); $x_{ijk}(t_0)$ is the initial condition of the restricted set of state variables. The notations of equations (18) may be shortened to:

$$X_{ijk}(t) = \phi_{ijk}(t,t_0)x_{ijk}(t_0) + \int_T \phi_{ijk}(t,\tau) B_{ijk}(\tau)z_{ijk}(\tau) d\tau , \quad (19a)$$

and

$$\Omega_{ijk}(t) = C_{ijk}(t)\phi_{ijk}(t,t_0)x_{ijk}(t_0) + D_{ijk}(t) z_{ijk}(t)$$
$$+ \int_T C_{ijk}(\tau)\phi_{ijk}(t,\tau) B_{ijk}(\tau) z_{ijk}(\tau) d\tau . \quad (19b)$$

As before, equation (6b), the important decomposition property of linear systems is expressed. The state determined behavior is

$$\Omega_{ijk}(t)_x = C_{ijk}(t)\phi_{ijk}(t,t_0) x_{ijk}(t_0), \quad (20a)$$

and the input determined behavior

$$\Omega_{ijk}(t)_z = D_{ijk}(t) z_{ijk}(t) + \int_T C_{ijk}(\tau)\phi_{ijk}(t,\tau)B_{ijk}(\tau)z_{ijk}(\tau)d\tau. \quad (20b)$$

In the case of a nonlinear system, equation (7), the state transition and response functions for the restricted set of diagnostic variables are, respectively,

$$X_{ijk}(t) = \phi_{ijk}(t,t_0)x_{ijk}(t_0) + \int_T \phi_{ijk}(t,\tau)B_{ijk}(\tau)\zeta_{ijk}[z_{ijk}(\tau), X_{ijk}(\tau)]d\tau \quad (21a)$$

and

$$\Omega_{ijk}(t) = C_{ijk}(t)\phi_{ijk}(t,t_0)x_{ijk}(t_0) + D_{ijk}(t)\zeta_{ijk}[z_{ijk}(t),x_{ijk}(t)]$$
$$+ \int_T C_{ijk}(\tau)\phi_{ijk}(t,\tau)B_{ijk}(\tau)\zeta_{ijk}[z_{ijk}(\tau),x_{ijk}(\tau)]d\tau. \quad (21b)$$

There is no separation of the response, equation (21b), into state and input determined components. Exogenous and endogenous contributions remain inextricable in the function ζ_{ijk}. The considerations for bioassays performed on nonlinear variables are therefore not the same as where linear variables are concerned.

2.3.2 Linearity vs. Nonlinearity in Toxicity Testing

Let the response of a set of diagnostic output variables on T in a controlled acute or chronic laboratory bioassay be, for the control, either (19b) (linear) or (21b) (nonlinear), and for the treatment:

$$\hat{\Omega}_{ijk}(t) = C_{ijk}(t)\phi_{ijk}(t,t_o)\,\hat{x}_{ijk}(t_o) + D_{ijk}(t)\,\hat{z}_{ijk}(t)$$

$$+ \int_T C_{ijk}(\tau)\phi_{ijk}(t,\tau)\,B_{ijk}(\tau)\hat{z}_{ijk}(\tau)d\tau \qquad (19c)$$

in the linear case, and

$$\hat{\Omega}_{ijk}(t) = C_{ijk}(t)\phi_{ijk}(t,t_o)\hat{x}_{ijk}(t_o) + D_{ijk}(t)\zeta_{ijk}[\hat{z}_{ijk}(t),\,\hat{x}_{ijk}(t)]$$

$$+\int_T C_{ijk}(\tau)\phi_{ijk}(t,\tau)B_{ijk}(\tau)\zeta_{ijk}[\hat{z}_{ijk}(\tau),\,\hat{x}_{ijk}(\tau)]d\tau \qquad (21c)$$

in the nonlinear case.

In (19b) vs (19c), and (21b) vs. (21c), if $x_{ijk}(t_o) \neq \hat{x}_{ijk}(t_o)$ and $z_{ijk}(\tau) = \hat{z}_{ijk}(\tau)$, then the environmental conditions are the same between control and treated variables and it is initial state that is manipulated. For example, if temperature, food, toxicant concentrations, etc. are the same, and organism densities or species compositions, etc. are different, then this corresponds to a state manipulation in the experiment. If $x_{ijk}(t_o) = \hat{x}_{ijk}(t_o)$ and $z_{ijk}(\tau) \neq \hat{z}_{ijk}(\tau)$, then it is the exogenous environment (temperature, toxicant, etc.) that is altered, with initial state held constant. Finally, if both $x_{ijk}(t_o) \neq \hat{x}_{ijk}(t_o)$ and $z_{ijk}(\tau) \neq \hat{z}_{ijk}(\tau)$, then both state and input variables have been varied between the experimental controls and treatments.

Recommendation 4. Regardless of the linearity or nonlinearity of test systems in toxicology experiments, great care must be taken to manipulate only those state or input variables which are the subject of testing. Inadvertent variations in nontest variables will contribute spurious components to the observed behavior, confounding or masking effects of the intended treatment, and making statistical discrimination of true results more difficult. The frequent death of control organisms in field testing performed by JRB and associates serves as a warning example. Control of test conditions by careful choice of subject populations, adequate replication, adherence to experimental design, etc. must be rigorous. "Quick and dirty" methodologies are forbidden as next to worthless.

The toxic response of test variables, reported, e.g., as a set of LC_{50} values, is the difference between treatment and control behaviors. In the linear case:

$$\Delta\Omega_{ijk}(t) = [\hat{\Omega}_{ijk}(t) - \Omega_{ijk}(t)]$$

$$= C_{ijk}(t)\phi(t,t_o)[\hat{x}_{ijk}(t_o) - x_{ijk}(t_o)] + D_{ijk}(t)[\hat{z}_{ijk}(t) - z_{ijk}(t)]$$

$$+ \int_T C_{ijk}(\tau)\phi_{ijk}(t,\tau)B_{ijk}(\tau)[\hat{z}_{ijk}(\tau) - z_{ijk}(\tau)]d\tau$$

$$= C_{ijk}(t)\phi(t,t_o)\Delta x_{ijk}(t_o) + D_{ijk}(t)\Delta z_{ijk}(t)$$

$$+ \int_T C_{ijk}(\tau)\phi_{ijk}(t,\tau)\,B_{ijk}(\tau)\Delta z_{ijk}(\tau)d\tau$$

$$= \Delta\Omega_{ijk}(t)_x + \Delta\Omega_{ijk}(t)_y. \qquad (22a)$$

For the nonlinear case,

$$\Delta\Omega_{ijk}(t) = [\hat{\Omega}_{ijk}(t) - \Omega_{ijk}(t)]$$

$$= C_{ijk}(t)\phi(t,t_0)[\hat{x}_{ijk}(t_0) - x_{ijk}(t_0)] + D_{ijk}(t)\zeta_{ijk}[\hat{z}_{ijk}(t)$$

$$- z_{ijk}(t), \hat{x}_{ijk}(t), x_{ijk}(t)]$$

$$+ \int_T C_{ijk}(\tau)\phi_{ijk}(t,\tau)B_{ijk}(\tau)\zeta_{ijk}[\hat{z}_{ijk}(\tau) - z_{ijk}(\tau), \hat{x}_{ijk}(\tau)$$

$$- x_{ijk}(\tau)]d\tau$$

$$= C_{ijk}(t)\phi(t,t_0)\Delta x_{ijk}(t_0) + D_{ijk}(t)\Delta z_{ijk}(t)$$

$$+ \int_T C_{ijk}(\tau)\phi_{ijk}(t,\tau)B_{ijk}(\tau)\zeta_{ijk}[\Delta z_{ijk}(\tau), \Delta x_{ijk}(\tau)]d\tau$$

$$= \Delta\Omega_{ijk}(\tau)_x + \Delta\Omega_{ijk}(t)_{zx}. \tag{22b}$$

These are the basic expressions for any toxic response observed under conditions i, with species mixture j, and chemical conditions k. The linear response is decomposable into portions due to initial state and input, $\Delta\Omega_{ijk}(t)_x$ and $\Delta\Omega_{ijk}(t)_z$, respectively, whereas the nonlinear response always carries with it terms, $\Delta\Omega_{ijk}(t)_{zx}$, in which state and input components are inseparable. This is the fundamental distinction between linear and nonlinear systems.

In the linear case, the state portion, $\Delta\Omega_{ijk}(t)_x$, captures the inherent response of the test system to the experimental treatment. This is intrinsic, and can be carried over as a property of the test organism(s) or other variables from one exposure situation to another (where i, j and k are, of course, the same). The input portion of the toxic response, $\Delta\Omega_{ijk}(t)_z$, represents the consequences of transferring the system to exposure conditions (with fixed i, j and k) where the driving exogenous inputs, z(t), differ. In the nonlinear case, the endogenous and environmental components of the observed toxic response cannot be separated, there will always be behavioral terms in the response of the form $\Delta\Omega_{ijk}(t)_{zx}$. In a subtle philosophical way, linear thinking can be established as the root cause of the experimental practice of placing test objects in a variety of environmental (input) circumstances in laboratory or field seeking to determine the role of environmental factors in dynamic behavior. For many, or most, biological systems, however, this is a role which cannot be isolated from the role of state.

2.3.3 Stationarity vs. Nonstationarity

Based on the results, $\Delta\Omega_{ijk}(t)$, tϵT, of bioassays, toxicant standards, σ_{ijk}, are arrived at through a variety of procedures and protocols. That is, for a set of toxicants and diagnostic variables, equation (10a) is established, where

$$\| \Delta\Omega_{ijk}(t) \| \leq \|\sigma_{ijk}\|. \tag{23a}$$

The standards are time invariant although, as indicated in Section 1.5, they could be time varying, $\sigma_{ijk}(t)$, tϵT, if necessary:

$$\| \Delta\Omega_{ijk}(t)\| \leq \|\sigma_{ijk}(t)\|. \tag{23b}$$

In expressions (23), $\Delta\Omega_{ijk}(t)$ are as defined in (22). These standards are then to be transferred to different sites, $i'\varepsilon E$, $j'\varepsilon S$, and $k'\varepsilon R$, at future times, $t'\varepsilon T'>T$, for the purpose formulated in expression (10) of protecting the subject environment, i.e., of assuring that

$$\| \Delta\Omega_{i'j'k'}(t')\| \leq \|\sigma_{ijk}\|. \tag{24}$$

There is a tacit assumption in this practice, probably based in linear thought which emphasizes intrinsic properties of organisms such as integrity through time, that time translation presents no problem, but it may.

Dynamical systems may be stationary (time invariant) or nonstationary (time varying). A stationary system is one which, begun in the same initial state at two different times, $x(t_o) = x(t_o')$, and subjected to the same subsequent stimuli, $z[t_o,t] = z[t_o', t']$, where $[t_o,t] = T$ and $[t_o', t'] = T'$ >T, will behave identically, $y[t_o,t] = y[t_o',t']$ at the two different times. Otherwise, $y[t_o,t] \neq y[t_o',t']$, a system is nonstationary. The existing EPA protocols do indicate the need to account for seasonal variability, and other time dependent sources of variation in toxic responses. The stationary property is essential to establish in order to move from bioassay to field without considering the time offset. Otherwise, care must be taken to establish that observations are made in comparable time periods in the life cycles of both the subject organisms and the eocsystems they occupy. Strictly, this is difficult to achieve, and the following recommendation is a statement of principle more than practice:

Recommendation 5. Stationary dynamics of subject systems, linear or nonlinear, cannot be assumed. Most natural systems are periodically time varying, however, with circadian, lunar or seasonal cycles in their behavior. EPA should strive to establish that their diagnostic output variables are stationary, but failing this, or if they are clearly nonstationary, then temporal controls should be exercised. Protocols to achieve this should be developed and applied to assure that only observations made during comparable phases of natural cycles are compared. Minimally, for example, summer data should only be compared to summer data, low flow data to low flow data, etc. Awareness of the importance of the stationarity problem is the first step in developing practical solutions to it, which EPA must develop as part of its site specific guidelines.

2.4 Monitoring

Environmental monitoring is the means of determining continued health,

$$\| \Delta y_{ijk}(t)\| \leq \|\sigma_{ijk}\|, \tag{10a}$$

or degradation,

$$\| \Delta y_{ijk}(t)\| > \|\sigma_{ijk}\|, \tag{10b}$$

of subject ecosystems based on establishing (24) or not through time. A spatial element is also present, and can be introduced by modifying equations (6) and (8). With $s\varepsilon\{s_1, s_2, s_3\}$, where s_1, s_2 and s_3 are the three dimensions of physical space, (5) and (7) become

$$\partial x_{ijk}(s,t)/\partial t = A_{ijk}(s,t)_1 x_{ijk}(s,t) + B_{ijk}(s,t)_1, z_{ijk}(s,t)$$

$$\partial x_{ijk}(s,t)/\partial s = A_{ijk}(s,t)_2 x_{ijk}(s,t) + B_{ijk}(s,t)_2 z_{ijk}(s,t) \qquad (25)$$

$$y_{ijk}(s,t) = C_{ijk}(s,t) x_{ijk}(t) + D_{ijk}(s,t) z_{ijk}(s,t)$$

for the spatially as well as temporally linear dynamical system, and

$$\partial x_{ijk}(s,t)/\partial t = A_{ijk}(s,t)_1 \, x_{ijk}(s,t) + B_{ijk}(s,t)_1, \zeta[z_{ijk}(s,t), x_{ijk}(s,t],$$

$$\partial x_{ijk}(s,t)/\partial s = A_{ijk}(s,t)_2 x_{ijk}(s,t) + B_{ijk}(s,t)\zeta[z_{ijk}(s,t), x_{ijk}(s,t]_2 \qquad (26)$$

$$y_{ijk}(s,t) = C_{ijk}(s,t) x_{ijk}(s,t) + D_{ijk}(s,t)\zeta_{ijk}\{\zeta_{ijk}[z_{ijk}(s,t), x_{ijk}(s,t)]_1,$$

$$\zeta_{ijk}[z_{ijk}(s,t), \, x_{ijk}(s,t)]_2$$

for the spatially and temporally nonlinear system. Other combinations include mixtures of spatial linearity and nonlinearity in the three spatial dimensions, for both temporally linear and nonlinear systems.

Monitoring may be conducted for several purposes, including (1) baseline surveys, (2) impact detection, (3) compliance monitoring, (4) establishment of causality, and (5) prediction. It is always limited to a restricted set of diagnostic variables, $\Omega_{ijk}(s,t) \subset y_{ijk}(s,t)$, whose necessary and sufficient relationship, (F3), to the full set of ecosystem variables, $y_{ijk}(s,t)$, must be assured:

$$\| \Delta\Omega_{ijk}(s,t) \| \leq \| \sigma_{ijk}(s,t) \| \Longleftrightarrow \| \Delta y_{ijk}(s,t) \| \leq \| \sigma_{ijk}(s,t) \|. \qquad (27)$$

Note here that a spatial as well as the temporal element has been introduced into the standards, i.e., $\sigma_{ijk}(s,t)$, for generality. This allows the option of permitting lowered standards [in general, higher values of the variables of $\sigma_{ijk}(s,t)$] in the plume from an outfall, or in downstream segments, if desired.

2.4.1 Baseline Monitoring
To establish nominal baseline conditions, some set of monitoring parameters, $\Omega_{ijk} \subset y_{ijk}$, is chosen. The variable $\Omega_{ijk}(s,t)$ represents a vector of these parameters measured in 1, 2 or 3-dimensional space, s, and discrete or continuous time t. Nominal conditions $\Omega_{ijk}(s,t)$ are established by measurement of the parameter set over appropriate intervals of space and time. The principal issues in this baseline monitoring are (1) choice of parameters (Section 2.2.1), (2) determination that they are necessary and sufficient to represent ecosystem behavior (Section 2.2.2), (3) spatial and temporal design of sampling, and (4) data assembly, analysis and presentation. Baseline conditions represent the starting point for all other forms of monitoring.

2.4.2 Impact Detection
In previous sections, toxic effects have been formulated as deviations from nominal, e.g., equations (22), and these deviations compared to standards expressed not as concentrations but, for dimensional consistency [see text ff. equation (9)], as effects. Two kinds of effects are chronic and acute. The latter may always be said to reflect standards expressed as concentrations. In chronic effects, however, concentration standards may never be exceeded yet long term, cumulative effects may occur. Examples include lowered dissolved

oxygen levels, impaired fecundity of organisms, stunted growth, anomalous age class distributions, high incidence of tumors, parasites or disease, etc. To monitor for these effects, some time and/or space integrated measure of deviational behavior may be required, e.g.

$$P[\Delta\Omega_{ijk}(s,t)] = \| \int_t \int_s \Delta\Omega_{ijk}(s,t) \ ds \ dt \|. \tag{28}$$

In impact detection monitoring, the subject system is considered perturbed (unprotected, F3b) when $P(\Delta\Omega_{ijk}) > \|\sigma_{ijk}(s,t)\|$, and unperturbed (protected, F3a) otherwise, $P(\Delta\Omega_{ijk}) \leq \|\sigma_{ijk}(s,t)\|$. The main issue in impact monitoring, in addition to those identified above for the baseline case, is defining suitable standards, $\sigma_{ijk}(s,t)$, that will reflect either acute or chronic effects. In the latter case, where impacts are cumulative and concentration based standards are never exceeded instantaneously or incrementally, integral measures such as (28) may have to be devised and effects based variables included in the standards set, $\sigma_{ijk}(s,t)$.

2.4.3 Compliance Monitoring

Determination of violations is a version of impact monitoring. The same elements are involved: (1) a set of monitoring parameters, $\Omega_{ijk}(s,t)$, (2) a standards set, $\sigma_{ijk}(s,t)$, incorporating both concentration and effects based measures of acute and chronic impacts, and (3) a sampling program temporally and spatially appropriate to the site. Compliance (F3a) is established by $\|\Delta\Omega_{ijk}(s,t)\| \leq \|\sigma_{ijk}(s,t)\|$ or $P(\Delta\Omega_{ijk}) \leq \|\sigma_{ijk}\|$, as appropriate, and noncompliance (F3b) otherwise.

2.4.4 Establishment of Causality

Except in simple cases where causality is obvious, as in oil spills, nuclear reactor accidents, or inadvertent releases of large quantities of toxic chemicals, the attribution of causality is a technically difficult problem. The basic formulation, expression (9), of the environmental protection problem in terms of general dynamical systems, equations (5-6) and (7-8), is an attempt to represent causality at a deep theoretical level. Causality is defined in terms of two properties, determinism [unique input-output relation, as provided by the basic dynamical system model], and nonanticipation (response to a stimulus not preceding in time the arrival of the stimulus). In general, the property of anticipation is not one found in ecological systems. For example, seasonal leads such as leaves turning color and falling before cold weather arrives have a mechanistic explanation in genetics and photoperiodism. Apparent anticipation is based on actually nonanticipatory, and determinate (dynamical), processes. The carryover of causally based output variables, $y_{ijk}(t)$, to a smaller diagnostic set, $\Omega_{ijk}(t)$, through necessary and sufficient conditional relationships between these two sets [expressions (10)] is a further effort to capture the causal relationship in the restricted set of variables by which an ecosystem is acutally observed.

A system is observed in monitoring to deviate, $\|\Delta\Omega_{ijk}(s,t)\| > \|\sigma_{ijk}(s,t)\|$ or $P[\Delta\Omega_{ijk}(s,t)] > \|\sigma_{ijk}(s,t)\|$, from its historically nominal condition soon after an anthropogenic activity is initiated in an area. Is the new activity responsible for the observed changes, or are these merely part of long term variability in nominal dynamics? Or more complexly, which ones of a set of human activities in an area are causing the deleterious changes that monitoring reveals, and which are not? How can causality be assessed? The means to do this is built into the principles implicit in Recommendations 1-4 above. By the considerations outlined therein, a

necessary and sufficient relationship is established between suspected cause and observed effect. Then, if the cause is present, the effect will be seen in monitoring data (sufficient), and if the cause is absent, the effect will not be observed (necessary).

2.4.5 Monitoring for Prediction

Predictive monitoring is also a technically difficult problem because it requires an offset in time. Observations made in earlier time, $\Omega_{ijk}(s,t)$, $t \varepsilon T < T'$, are used to project conditions in future time, $y_{ijk}(s,t')$, $t' \varepsilon T'$. In terms of conditional logic, prediction requries that

$$\|\Delta\Omega_{ijk}(s,t)\| \leq \|\sigma_{ijk}\| \overset{(1)}{\Longleftrightarrow} \|\Delta\Omega_{ijk}(s,t')\| \leq \|\sigma_{ijk}\| \overset{(2)}{\Longleftrightarrow} \|\Delta y_{ijk}(s,t')\| \leq \|\sigma_{ijk}\| \quad (29a)$$

or

$$P[\Delta\Omega_{ijk}(s,t)] \leq \|\sigma_{ijk}\| \overset{(1)}{\Longleftrightarrow} P[\Delta\Omega_{ijk}(s,t')] \leq \|\sigma_{ijk}\| \overset{(2)}{\Longleftrightarrow} P[\Delta y_{ijk}(s,t')] \leq \|\sigma_{ijk}\|$$

$$(30a)$$

and

$$\|\Delta\Omega_{ijk}(s,t)\| > \|\sigma_{ijk}\| \overset{(1)}{\Longleftrightarrow} \|\Omega_{ijk}(s,t')\| > \|\sigma_{ijk}\| \overset{(2)}{\Longleftrightarrow} \|\Delta y_{ijk}(s,t')\| > \|\sigma_{ijk}\|$$

$$(29b)$$

or

$$P[\Delta\Omega_{ijk}(s,t)] > \|\sigma_{ijk}\| \overset{(1)}{\Longleftrightarrow} \|\Delta\Omega_{ijk}(s,t')\| > \|\sigma_{ijk}\| \overset{(2)}{\Longleftrightarrow} \|\Delta y_{ijk}(s,t')\| > \|\sigma_{ijk}\| .$$

$$(30b)$$

Necessary and sufficient conditions (1) are based on Recommendation 5, establishment of the stationary property, and conditions (2) are based on adequacy of the relationship between system variables y_{ijk} and diagnostic variables Ω_{ijk}, as developed in Recommendations 1-4. Strict adherence to the principles inherent in these recommendations should make a predictive monitoring methodology theoretically possible, but discovering the protocols required to gain this strict adherence is no easy technical matter.

2.4 The EPA Protocols

This section will endeavor to briefly examine EPA's approaches to environmental protection against chemical toxicants in context of the foregoing theoretical specification of the basic problem.

2.4.1 The National Guidelines

The national criteria are global in their intent to deal with all ecological conditions, $i = E$, biological species, $j = S$, and resources, $k \varepsilon v \varepsilon R$ (see Section 1.5). That is, they are designed to protect all aquatic environments:

$$\| \Delta y_{ESR}(t)\| \leq \|\sigma_{ESR}\| , \qquad\qquad (31)$$

where σ_{ESR} is the national standard for the k'th toxicant. The impossibility of achieving such a global solution represented by (31), without making the standards unrealistically stringent, rightly leads to the site specific approach. The theory laid down here at least verifies that, in principle, the site specific approach is the correct one.

The national standards σ_{ESR} fail to implement (31) effectively for a variety of reasons. In attempting to protect all environments, they in effect protect no environments with significant problems. They deviate strongly from all the recommendations provided above:

Recommendation 1. The national standards are based solely on toxicology data; ecological considerations are ignored. Biotic variables only are considered; fast-time abiotic ones are needed. An effort is made to account for both chronic and acute effects, and to aquire invariant information on representative sets of (animal) species. Site specific choices of species are, of course, ignored.

Recommendation 2. No effort is made to establish necessity or sufficiency of toxicity information in the national data set.

Recommendation 3. By definition, the national standards are non-site specific.

Recommendation 4. The stringent control requirements of laboratory toxicology experiments have probably, in most cases, not been met.

Recommendation 5. Stationary dynamics of ecological systems are assumed by the fact that the stationarity problem is ignored.

Recommendation 6. The national guidelines, which are to be used in lieu of site specific information, should be reviewed and improved with respect to the principles inherent in the preceding recommendations which apply.

2.4.2 Site Specific Rationale

Site specific guidelines are designed to introduce local considerations into the process of deriving standards. As previously formulated, expression (9), the objective is

$$\| \Delta y_{ijk}(t)\| \leq \|\sigma_{ijk}\| , \qquad\qquad (9)$$

where $i \subset v \varepsilon E$, $j \subset v \varepsilon S$ and $k \subset v \varepsilon R$. In this, it is recognized (1) that the combination j of species at a site (with characteristics i and resources k) may be more or less sensitive than those used in the national criteria data set, or (2) that the water quality characteristics, i and k, at that site may alter the toxicity of the species collection j to the chemical of interest in k. Site specific criteria are designed to deal, singly and jointly, with these conditions, and in addition, to account for seasonal variations in water quality.

Formulation (9) exactly reflects this rationale. EPA's definition of "site" (e.g., EPA 1982, p.3-4) is adequately captured by the concepts $i \subset v \varepsilon E$, $j \subset v \varepsilon S$ and $k \subset v \varepsilon R$ as unique combinations of, respectively, ecological conditions, species, and resources. However, the assumption (EPA 1982, p. 3-6) that species sensitivities and toxicological effects derived from laboratory tests will be similar to those in the field is at variance with fundamental considerations in the role of environmental factors in system dynamics. There is little in theory to justify such an assumption (Section 2.3); if it were true, then by the same rationale the results from any

particular toxicology experiment in the laboratory or field could be extrapolated to the national level, and there would be no need for site specific guidelines. To the contrary, "biological integrity" is frequently judged to prevail in situations where toxic concentrations exceed standards, and biodamage may occur under conditions where concentrations chronically never exceed standards (EPA 1982, p. 3-7).

Four procedures have been developed to implement the site specific rationale. They are the recalculation, indicator species, resident species and heavy metal speciation procedures.

2.4.3 Recalculation Procedure

This procedure is designed to account for differences in sensitivities of resident species, $j \subset v \varepsilon S$, to a toxic chemical for biological reasons. "Resident species" are defined, acceptably, as those which normally occur at a site during a time interval T which spans seasonal variability (EPA 1982, p. 3-12). Extinct species at the site are not included, nor do long term variations in species lists appear to be taken into account.

The recalculation procedure permits eliminating families required to establish the national acute toxicity standards. Defects in the strictly taxonomic approach to selecting diagnostic biota, acknowledged in the sensitivity of final acute values to family selections (EPA 1982, p. 3-21), have already been pointed out (Section 2.2.1). The recalculation procedure, being tied to this approach, shares this fundamental flaw.

In addition, there are specific problems. A site specific acute standard, $\sigma_{ijk}^{(a)}$ (the final acute value, FAV), is calculated after deleting nonresident species from the list of those used to determine the national standard, σ_{ESP}, and meeting minimum data set requirements. This site specific standard is then arbitrarily adjusted for conservativeness to obtain a site specific maximum instantaneous concentration, $\sigma_{ijk}^{(a)}/2$. This is scientifically baseless, although justified by the generally unrefined character of the approach. A final chronic value, $\sigma_{ijk}^{(c)}$, is obtained by applying a laboratory based acute-chronic ratio to the FAV. This violates several basic principles (Sections 2.2.3 and 2.3) pertaining to the extrapolation to field situations from laboratory data.

Recommendation 7. The recalculation procedure is inadequate, both basically and in terms of technical details, to account for species based sensitivity differences between site specific toxicity responses and responses from which national standards are derived. The problem should be restudied by EPA, and a scientifically better grounded protocol (Section 2.2.1) for selecting biotic diagnostic variables formulated. Many of the specific considerations developed in the recalculation procedure can be carried forward for improved elaboration in a more definitive methodology.

2.4.4 Indicator Species Procedure

This method is used when site water quality affects the toxicity of a compound. That is $\sigma_{ijk} \neq \sigma_{i'jk'}$, where $i' \neq i$ denotes ecological factors such as pH, hardness, alkalinity, carbon dioxide equilibrium relationships, salinity, etc., and $k' \neq k$ denotes resource factors such as organic solutes, inorganic and organic colloids, and suspended particles. The method assumes no difference in response of resident biota from those species in the national data base. It uses a simple multiplier, the water effect ratio, to correct for differences between site water and laboratory test water. The water effect ratio is calculated using resident species at the site, or "acceptable" (EPA 1982, p. 3-22) indicator or surrogate species. As in the recalculation

procedure, a final acute value, $\sigma_{ijk}^{(a)}$, is calculated and then conservatively adjusted to a maximum instantaneous concentration, $\sigma_{ijk}^{(a)}/2$. Final chronic values, $\sigma_{ijk}^{(c)}$, are computed either from the FAV, or chronic toxicity testing with any (resident or nonresident) fish and invertebrate species.

The indicator species procedure is positive in recognizing that ecological, $i \subset E$, and resource, $k \subset R$, factors may modify the toxicity responses or bioaccumulation characteristics of site specific resident species, $j \subset S$ (Section 2.2.3). However, in effect, this recognition itself negates the narrow toxicology based philosophy of EPA which otherwise generally ignores ecological considerations (e.g., Section 2.2.1). Specific criticisms of the indicator species procedure are: (1) the assumption, in order to use it, that no species response differences exist between resident and national data base species, which is unrealistic; (2) "acceptability" for nonresident indicator or surrogate test species is not defined; (3) apparently, any two species of a fish and an invertebrate may be used in testing for chronic toxicity; and (4) in an effort to be practical, the procedure is in final analysis too simplistic.

Recommendation 8. The indicator species procedure, as it presently exists, is inadequate as a method. However, many of the principles inherent in it should be retained in a totally redesigned approach to incorporating ecological considerations more fundamentally into the EPA site specific protocols, in accordance with Recommendations 1-3 (Section 2.2).

2.4.5 Resident Species Procedure

This procedure is to be used when there is reason to suspect that both species and water quality differences may cause differences in toxicity or bioavailability of a chemical, i.e., $\sigma_{ijk} \neq \sigma_{i'j'k'}$, where $i \neq i'$ and $k \neq k'$ denote environmental differences, and $j \neq j'$ denotes species differences. The procedure calls for applying both the recalculation procedure to account for species differences, and the indicator species procedure to account for water differences.

Recommendation 9. The resident species procedure is philosophically consistent with previous recommendations which emphasize the need to incorporate both ecological and toxicological desiderata into a site specific methodology. While both the recalculation and indicator species are individually philosophically defective, together they combine to remove this criticism, and basically only technical flaws remain. Therefore, the resident species procedure should be retained as the cornerstone of an interim site specific methodology, while EPA moves forward to develop definitive procedures that are both philosophically and technically matched, in realistic pragmatic ways, to the difficult requirements of the problem.

2.4.6 Heavy Metal Speciation Procedure

The national standards for metals are expressed as total recoverable metal based on laboratory data on total recoverable, or acid extractable metal concentrations. Metals exist in a variety of forms, each with specific toxicological characteristics, i.e., $\sigma_{ijk'} \neq \sigma_{ijk}$, where $k' \neq k$ denotes two different forms of a metallic element in question. In setting site specific standards for methods, either the indicator or resident species procedures may be used to modify the national standards.

Recommendation 10. Based on the general inadequacy of the indicator species procedure by itself, the resident species procedure should be employed as an interim methodology to set site specific standards for heavy metals. However, a more definitive metals protocol should be developed, in accordance with the principles outlined in this report.

3. SUMMARY AND CONCLUSIONS

The problem of setting water quality standards for toxic substances is one of the most difficult applied environmental problems possible because (1) it intersects head-on biological and environmental diversity and variability, and (2) requires a refined integration of all this complexity mapped into a quantitative "standard" for each toxic material. The problem pushes environmental science well beyond the current state-of-the-art.

As presented to this committee, the site specific problem was poorly specified. Underlying philosophical and logical issues had not been systematically examined as a basis for the pragmatic methodologies that had to be developed and thus took precedence. As a result, the existing procedures are deeply flawed and in need of revision. The fundamental wisdom to move from national to site specific standards is unchallenged, however, and is endorsed as correct and urgently needed.

With the need to bring the problem into better philosophical and logical focus, a preliminary attempt has been made here to provide a general systems theory specification that can underly future methodological development. Ten recommendations have been made which encompass the principles exposed by this theoretical development:

1. Diagnostic variables should be carefully chosen to include both biotic and abiotic ecosystem properties which reflect ecological as well as toxicological considerations;

2. Necessary and sufficient relationships should be established between the restricted set of diagnostic variables and the larger set of all relevant ecological variables;

3. Standards should be site specific, and matched as closely as possible to the conditions and species at each site;

4. Laboratory toxicology testing must be conducted under carefully specified and stringently controlled conditions;

5. Protocols should take account of temporal variability in toxicant effects.

6. National guidelines, if they are to continue to be used in lieu of site specific information, should be improved in accordance with the preceding recommendations;

7. The recalculation procedure should not be used by itself;

8. Neither should the indicator species procedure be used by itself;

9. The resident species procedure should be used as an interim methodology until better ones have been developed; and

10. The heavy metal speciation procedure should be based on the resident species procedure until improved methodologies are developed.

Finally, a general recommendation based on all the preceding material may be offered in conclusion:

SUMMARY RECOMMENDATION. EPA should (1) further develop better specification, in theoretical terms, of the environmental protection [Section 1 and expression (9)] and protocol [expression (10) and Section 2] problems, building on the start made here, and (2) employ the continually improving, and possibly alternative, formulations of these problems to develop better site specific methodologies that are pragmatic, consistent with emerging principles, and conformable with both ecological and toxicological criteria.

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APPENDIX 2

EVALUATION OF CASE HISTORIES IN RELATION TO

FIELD VERIFICATION OF PROPOSED GUIDELINES

Field testing for the development of site-specific criteria was
completed at 19 sites in 16 States. Sites, pollutant sources,
and types of pollutants were selected by EPA regional offices
and the States. Most of the field work was contracted out to
JRB and Associates, with the exception of studies in California
related to the use of 2,4-D esters in forest silviculture.

The work was developed to test a wide variety of locations,
types of aquatic habitats, and chemical pollutants. These
included pesticides, heavy metals, ammonia, and chlorine from
wastewater treatment plants.

Field studies were based on the Guidelines for Deriving Site-
Specific Water Quality Criteria for the Protection of Aquatic
Life and Its Uses. The case histories did contain much useful
information, but as a whole the Committee found all of them to
be inadequate to support site-specific modifications of the
criteria.

The lack of specificity for the development of field studies,
including such items as collection and treatment of samples,
appropriate selection of test species and more specific
information concerning the selection of actual sites to be
sampled and the conditions under which the sampling should be
done led to some problems with the design and conduct of field
studies. Most of the studies involved the testing of the
effects of a single pollutant from a point source of pollution.
One field test, completed in California, was of a non-point
source of pollution, the aerial application of the herbicide
2,4-D esters.

The initial selection of sites was dictated by the location of
the point source of pollution and by the order of entry of
other sources of pollution in addition to the potential for
dilution of the initial sources. Generally, sites were
selected upstream from the sources of pollution, at or near
the point source, and in additional areas in the recovery
zones. The protocol was to sample water from upstream sites,
at the point source, and then at various locations in the
plume and in the recovery zones. Upstream water was used to
test resident species to develop water quality ratios for the
specific pollutants studied. The selection of sites seemed a
matter of convenience rather than a deliberate choice to make
locations comparable in slope, habitat characteristics, and

2-1

other water quality measurements. In some of the studies,
the selection of comparable sites may have been difficult
because bottom sediments, aquatic plants, and characteristics
of the stream or river were altered by the source of pollution.
This was true for those point sources that included heavy
sediment loads which altered the character of the stream or
river bed. In some instances, there might have been a better
choice of more comparable sites for comparing water quality
and species composition.

The selection of species to be tested presumably was based on
the available information on resident species and in some
cases the initial collections of organisms from the site itself.
Some of the studies actually used species collected from the
site, while others utilized animals from other sites, such as
rivers, lakes, ponds, and, in some instances, fish from
hatcheries were used in the pollutant testing for water quality
ratios. It would seem most appropriate to use organisms
collected in the actual waters where the testing was to be
done; this would insure a more natural response to the test
pollutant in both static and flow through systems. Response
of organisms taken from other waters and from hatcheries would
undoubtedly lead to greater variance in response to site water
and an additional variation to the added pollutant based on
prior exposures to water qualities of unknown characteristics.
The purpose of the site-specific water quality criteria would
seem to be related to the actual response of resident species,
and more effort should have been made to test actual residents
from the body of water being used as a field site. If suf-
ficient numbers of organisms were not available at the
appropriate time or were not convenient to being collected, a
greater effort should have been made to collect organisms from
similar bodies of water nearby; it would have been preferable
to use species from the field rather than hatchery-reared
organisms.

The development of the field testing protocol should have been
carefully monitored in terms of the quality of the water being
used for the static and flow through tests. Measurements of
water quality at the intake source should have been compared to
that in the tanks at the time of testing. Collection, pumping,
and storage of water, even for short periods of time, could
have altered these parameters to where they may have influenced
the response of the organisms to the pollutant being tested.
In some of the studies, high losses of control animals did
indicate that water quality characteristics were not
sufficiently beneficial to the continued existence of the
animals being tested. These problems in some of the studies
negated the results of the pollutant test.

Surveys of organisms at the various sites was done by a
variety of methods, and the efficiency of these techniques was
not always taken into consideration. In some instances, the
use of shockers or seines would produce varied results based
on water quality characteristics as well as total flow and
velocity. Development of diversity indices would not be useful
if variation in sampling techniques biased the collections.
In one instance, high flow levels precluded adequate surveys,
yet the field work continued despite adverse conditions.

Quality control in the analytical work seemed to be appro-
priate, but the addition of the toxic metals to waters and
their subsequent assay may have to be revised in the light of
new information related to metal speciation. Actual testing
of species in laboratory, reconstituted, and flow through
waters varied. In some cases, loading factors varied; in some
tests, species were mixed in flow through waters; in others
species were tested separately; in other cases water was
aerated; and in other instances it was not disturbed. This
may have been related to the pollutant being tested in terms
of volatility; but, in total, the test conditions could have
been better standardized.

Prior exposures to toxic substances for resident species may
present a problem in testing programs. Body burdens of various
substances could result in resistance or greater susceptibility
to the test pollutant as a result of synergism or alteration in
physiological conditions. Handling stress, particularly of
hatchery fish, might also alter susceptibility to the test
substances. Test organisms that display resistance should be
tested in laboratory water to determine if water effects ratios
are causing the altered effect of the pollutant, or whether
developed resistance may have occurred in the resident species.
In some tests the sources of species came from all different
sites, from lakes rather than stream systems and from hatcheries
rather than the field. Water effects ratios for invertebrates
and vertebrates that are diametrically opposed should be
discussed and perhaps analysed further. It would seem that
actual residue data from the resident species would be valuable
in assessing the results from laboratory exposures.

All of the studies were of short duration, presumably as a
result of time and cost restrictions imposed on the field
testing program. Some prior development of background materials
for the various sites would have been useful in terms of stream
gradients, habitat types, stream, flow characteristics,
availability of test species from the site, and the potential
added sources of pollutants at both upstream and downstream

locations. Historical data on flow rates might have led to
more opportune times to sample and perhaps a more equitable
evaluation of the effects of the pollutant load on the resident
aquatic species. The use of resident species taken from the
recovery zone may be questionable because the potential for
some prior exposure would be increased. Physical factors and
flow conditions could have direct impacts on pollutant effects
and should be included in all study reports.

Field studies should develop some perspective of the pollutant
problem in relation to the site, the pollutant, water uses, and
potential effects of alteration in the standards. This was
most appropriately done in the California study on the herbicide
2,4-D esters.

The field studies have shown that a more detailed protocol
is desirable. Strict guidelines may not be applicable to all
sites and for all species being tested. Required testing of
water from input sources, in the collection systems, and in
the laboratory tests should be standardized. The selection of
test organisms should, if at all possible, be from the actual
site in waters free from the test pollutant. More attention
should be given to the prior description of the test site; and
the actual water quality, habit types, flow conditions, and
pollutants should be available prior to the actual site selection
and field testing. This would allow for the development of a
more efficient and desirable testing program both in time and
performance. Additions of pollutants, particularly the
heavy metals, should be done according to the latest methodology.
Field testing reports should also contain some review of
potential decisions concerning pollutant loads, and they should
also recommend a plan for monitoring the site following any
change in the site-specific standards for a toxic substance.

Specific Comments on Case Histories

The following pertains mainly to the JRB reports or contracted
studies, but the principles, nevertheless, apply all around.
There are two outstanding aspects involving philosophy/
approach/principles that are generally applicable. They cannot
be solved by toxicological tests or improvement of bioassay or
chemical analytical techniques.

The first is best expressed on page 41 of the Mill Creek, Clinton, Iowa draft report. The last paragraph makes the point that the acute and chronic toxic tests are not telling the story of the effects upon organisms. A quote--from context to be sure--in mid-paragraph states this cogently, "Total taxa, biomass, chlorphyll, and diversity showed a continual decline at downstream station which was not consistent with measured toxicant concentrations." From this, together with what we have learned about the inaccuracy of both the derived and experimental determination of toxic criteria, it is probable that the pursuit of these sorts of data alone are a loss of effort and money and have solved no problems, nor is continued pursuit of this course bound to solve the problems at hand.

What is needed is a basic understanding of ecosystems and what influences changes that they undergo. It is also essential that they be recognized as multidimensional and that toxic, lethal, and population altering effects cannot be measured by LC$_{50}$'s alone. The amount of data required or type of understanding needed is really no more or less than that which would be required for the present techniques, ideally. What _is_ required is a new direction in, or approach to, thinking as it concerns ecosystems by the authorities enforcing the Clean Water Act.

The second aspect is closely related to this and involves an attitude toward sampling. Almost all of the contractors fall back on "Standard Methods" for their techniques in sampling and also in the selection of locations for sampling. Then they appear to be amazed when these methods do not work for their particular situation. Often, also, location of stations and time of sampling are not carefully done with respect to knowledge of the site. The best all around approach to such problems is to decide what it is one needs to know, the feasibility of getting this information, how the site (system) in question can best reveal this information, and what sorts of techniques can _best_ be employed, standard methods notwithstanding. This approach should be taken first and foremost _after_ determining what ecosystems are involved, what is already known about them, and what more needs to be known to decide on an assay action. Only a systematic approach can reveal useable answers.

The following pertain to more specific issues:

Mill Creek, Clinton, Iowa
p.9. Setting up artificial periphyton substrates was a good idea, but why were they floated in mid-water? Why were they

2-5

not put on the bottom or sides of the creek where the
periphyton occur in greater abundance?

p.9.   If there is a question of dissolved oxygen vs. chemical
or biochemical oxygen demand in various locations, why were
diurnal and turbulence tests not run to determine the oxygen
source?

The use of periphyton as an index of growth stimulation
or toxicity, i.e., chemical influence on the ecology, is a
good one, but the investigators need to have a better
understanding of both the ecology of these types of organisms
and how the systems interact.

Iowa River, Marshalltown, Iowa
p. 2-7.   The periphyton methodology needs rethinking, also it
should be realized that some of these organisms are not only
indicators but species of importance in their effects upon
other organisms.   This is a point that many of the investi-
gations seem to bypass.

p. 2-8.   When there is a distinct peculiarity in response,
e.g., mayfly to ammonia toxicity, one doesn't just say that
the LC_{50}'s could not be calculated; one tries to find an
explanation for the behavior (e.g., possible internal pH
change or some binding of NH_3).

p. 2-6, 2.2.3.   One fish collection may not necessarily
characterize an area.

2.2.4.   Physical characteristics of the river should reveal
areas of concentration of forms (e.g., where some stage of
early development occurs).

pp. 3-4.   A coarse quantitative net, possibly like a plankton
net, can be used for capture and, if intelligently designed,
can be used at various depths, at least for relative number of
organisms determinations.

Selser's Creek, Ponchatoula, Louisiana
p. 2-5.   The floating macro-plants should have been sampled
quantitatively.   The substratum furnished may have been
the principal source of organisms.

p. 2-6.   The turbidimeter does not give the best estimate of
visibility, light penetration, or supended particles.
Newer techniques are no more trouble or expense and far
more revealing of usable information.

The contractors are using a good set of indices when applied
with logic (top of the page).

p. 4-1. The characteristics discussed in the first paragraph
also probably influence distribution of toxic substances
physically and chemically by substrate binding.

Also, a different approach to alkalinity anions is needed
since there are low pH's; also the hardness should not be
calculated as $CaCO_3$ under these conditions but rather as
meq. It should be determined what are the important anions
in the water at these pH's.

The contractors show different taxa at different stations.
What are the effects of ambient and toxic conditions on the
distribution of indigenous forms?

Since this is an important farm area, what testing has been
done on PO_4, soil quality, and other non-point source
intermittent changes?

(Lindane) Salt Creek, Lincoln, Nebraska
By their own admission, the non-point sources are important
contaminators. If so, break-through concentrations plus
buildups may be found frequently in several local areas with
probable effects on wells and groundwater. A regime of
sampling to accommodate non-point sources should be
established.

There should be a whole new approach to benthic organism
study methods. Organisms occur where they are, which may
be on the bottom, in suspension, temporarily in suspension,
dirunally active, etc., and sampling methods should be based
on these occurrences, not on what bottom sampling techniques
are available in "standard methods," which do not apply to
all types of physical situations.

Flint River, Near Flint, Michigan
Here, some attempt has been make to use other dimensions,
such as time and seasonality, but this has not been well
thought out, e.g., seasonality is not told by the month, but
rather by the change in conditions; and time effects are
often really what happens in the darkness, under different
sun angles, and under cloud cover.

Hutchinson, Minnesota
In general, there needs to be improvement in field sampling techniques and possible mathematical relationships of the behavior of organisms to field characteristics, e.g., speed of currents, particle suspension physics, bottom types, and the seasonal changes in these. Plankton net type of instream capture should also be considered.

There should also be concern with a variety of non-point source distributed substances, since this is an agricultural area.

p. 3-8. If nutrients are an important factor in this area, standard microplant culture assays should be instituted using both indigenous species and EPA-accepted microplants in stock culture.

Boggy and Skeleton Creeks, Enid, Oklahoma
This survey should definitely have included diurnal studies on both dissolved oxygen and other factors. They also have a non-point source problem, since this is farmland.

Mingo Creek, Tulsa, Oklahoma
p. 2-10. Samples were taken from only the riffle areas.

p. 3-2. Hardness and alkalinity study may have been very important here, especially at night.

p. 3-12. This was a clear demonstration of physical factors influencing population types and trophic levels. It is too bad that the study was not pursued a little further, as it might have given more insight into heavy metal distribution, into chemical species and chemical activity. Seasonality is also very important with regard to stream concentration.

California Fish and Game
What is the normal equilibrium in natural waters of 2,4-D esters and acid and what are the factors controlling them?

California State Water Resources Control Board (2,4-D esters)
p. 9. There is obviously a physiological difference between rainbows and steelheads. What is this difference due to? What are the conditions of 2,4-D ester hydrolysis?

p. 17, end of paragraph 1. Is the Committee to understand that a sampling device that is automatic or activated from shore cannot be developed and be cheaper in the long run that the approach currently used?

p.31. Again this points toward the need for methods in non-point source contamination detection.

p.32, last paragraph. This points up the need for survey planning to get the _kind_ of results that are useful and to use pooled resources.

These people have severe non-point-source-in-remote-areas problems. Therefore, special considerations should be given for subsurface flow regimes, soil surveys, etc., at key locations along with either automatic or composite (pooled) sample collecting gear. Much of this can be done at low cost.

Pullman, Washington (Department of Civil Engineering)
p.12. More needs to be known about the physiological behavior of the sculpin. How was the natural water filtered before use?

p.13 and 15. Either better handling techniques need to be found for ephemerids or work should just be done on stoneflies. Possibly, collection of the organisms should be done at a stage when they might be more hardy.

p.20. This shows the potential value of behavior as an index of toxicity in both the field and laboratory (Drift of ephemeroptera out of streams dosed with copper--see also near top of p.21.)

p.i. There are indications of concentration of algae.

p.ii. (Of the abstract from Funk et al.) The indications here are that physiological clearance tests of animals show the importance of function (clearance) in the assay of higher organisms. This is not even suggested in any of the other surveys.

Camp, Dresser, and McKee, Hamburg, New Jersey
p.2. Why was dilution water not obtained above the Ames plant? What about runoff from the highway?

p.4. Last paragraph. How does mixing, effluent, and river level vary seasonally?

p.8. If coefficients of variance were run, there may have been differences between lab and site water.

p.9. The answer to the posed problems might be in the seasonal change in stream characteristics.

p.10, last paragraph. It is not clear whether no further tests are recommended because of expense. The fate aspect of the contaminant may be stream mitigation or concentration in the stream wall or sediments until there is a break-through to the stream horizontally or to groundwater vertically.

Carolina Mirror
With the data obtained on the UT and associated streams, one would have thought that more work might also have been done on sediments and on physical characteristics of the stream shape and flow. Also, seasonal input would be necessary to know whether or not the carefully described characteristics change throughout the year.

Hempstead Wastewater Treatment Plant, Maryland
Seasonality, rainfall, pH influence on the sediments (binding of ammonia) show the need for some checks thoughout the year. The same is true at Buford, Georgia, and in the latter place as well, the binding of NH_3 to waste particles and carbanino complexing can change the degree of toxicity at least on micro-organisms.

Finally, the Committee would like to recommend two papers that bear strongly on the types of assays we have been reading. One deals with bioconcentration by rainbow trout and the other with the important and definite distribution of fishes in habitat gradients along stream lengths.

Oliver, Barry G. and Arthur J. Niimi (1983). Bioconcentration of chlorobenzenes from water by rainbow trout: Correlation with partition coefficients and environmental residues. Science and Technology 17(5):267-291.

Schlosser, Isaac J. (1983). Fish community structure and function along two habitat gradients in a headwater stream. Ecol. Monogr. 52(4):395-414.