DEVELOPING AN ECOSYSTEM-BASED CAPABILITY FOR ECOLOGICAL RISK ASSESSMENTS

John Cairns, Jr.
Paul V. McCormick
Virginia Polytechnic Institute and State University

Abstract. Ecological risk assessment is a scientific process for estimating, with a known degree of certainty, anthropogenic effects on the integrity of natural ecosystems and the services they provide. An inordinant focus has been placed on bottom-up risk assessment strategies, which emphasize laboratory-based testing. Laboratory protocols alone are incapable of estimating uncertainties associated with attempts to extrapolate data among levels of environmental complexity, biological organization, and a multitude of potential impact scenarios. Top-down approaches to risk assessment, which utilize ecological indicators present in natural ecosystems, can address these problems and ensure that previously-stated environmental objectives are met. Monitoring programs designed to protect ecosystem integrity should include: (1)compliance indicators—for assessing the degree to which previously-stated environmental conditions are maintained; (2)diagnostic indicators—for determining the cause of deviations outside the limits of acceptable conditions; and (3)early warning indicators—for signalling impending deleterious changes in environmental conditions before unacceptable conditions actually occur. While scientific, economic, and political constraints preclude development of an ideal monitoring system at this time, implementation of less comprehensive programs currently is feasible.

INTRODUCTION

The nature and extent of human impact on the environment has changed drastically since the dawn of the industrial revolution. The third decade is beginning of what could be termed a period of environmental awakening, during which increased public concern over environmental damage and human health has led to several important regulatory mandates for environmental protection (e.g., the Clean Air Act of 1970, the amended Federal Water Pollution Control Act of 1972, and the Toxic Substances Control Act of 1976). Many tangible benefits have resulted from these and other pieces of legislation. However, environmental problems continue to become more diffuse and complex and, therefore, more difficult to confront. On a local and regional scale, this trend is exemplified by a shift in concern from point source discharges of toxic materials to the less tractable problem of nonpoint source pollution. On a larger scale, the global depletion and destruction of natural resources and the prospect of ozone depletion, climate change, and other global concerns generated by human activity threaten to alter the biosphere and civilization, as humans now know them. There is little precedent to the difficult choices currently facing society with regard to environmental protection and restoration. Measures taken to avert an emerging global environmental crisis will depend largely on the perceptions of socioeconomic and environmental consequences of alternative actions, as determined by means of risk assessment.

What are the potential hazards posed by persistent and emerging environmental problems to natural ecosystems, and what degree and types of prevention and mitigation are necessary to avoid unacceptable damage and maintain natural ecosystems as self-maintaining systems capable of providing essential services to society? Ecological risk assessment protocols should provide scientifically justifiable information to answer these questions with a known degree of

Dr. John Cairns, Jr. is University Distinguished Professor of environmental biology and Director of the University Center for Environmental and Hazardous Materials Studies at Virginia Polytechnic Institute and State University, Blacksburg. He also is a Certified Environmental Professional with the National Association of Environmental Professionals. Among his awards are Fellow, American Association for the Advancement of Science in 1969; the Presidential Commendation in 1971; the Charles B. Dudley Award in 1978 for excellence in publications from the American Society for Testing and Materials; Superior Achievement Award, U.S. Environmental Protection Agency in 1980; the Founder's Award of the Society for Environmental Toxicology and Chemistry in 1982; the Icko Iben Award from the American Water Resources Association in 1984; the B.Y. Morrison Medal in 1984; Fellow, American Academy of Arts and Sciences in 1988; the United Nations Environmental Programme Medal in 1988; the American Fisheries Society Award of Excellence in 1989; the 1991 Life Achievement Award in Science by the Commonwealth of Virginia and the Science Museum of Virginia; and member, National Academy of Sciences, 1991.

Paul V. McCormick is assistant professor of biology at Elizabethtown College and an adjunct professor in the University Center for Environmental and Hazardous Materials Studies at Virginia Polytechnic Institute and State University. Dr. McCormick has conducted research in several areas of environmental science and ecology, much of which has emphasized aquatic microbial communities. He has published in both national and international peer-reviewed journals and has served as a consultant to industry and the government.

Requests for reprints should be addressed to John Cairns, Jr., 1020 Derring Hall, University Center for Environmental and Hazardous Materials Studies, Virginia Polytechnic Institute and State University, Blacksburg, VA 24061-0415.

THE ENVIRONMENTAL PROFESSIONAL Volume 14 pp. 186-196 1992.

0191-5398/92 $3.00 + .00

certainty. Depending on the overall objective of a particular risk assessment program, this information should be useful for making decisions regarding management priorities and regulatory guidelines at relevant spatial scales (e.g., ecosystem, ecoregion, or biosphere). Most importantly, information gained from ecological risk assessment should be relevant to the process of risk management, which weighs not only scientific evidence, but also socioeconomic and political concerns, in determining what level of risk is acceptable to society as a whole.

An ecological risk assessment begins with the assumption that certain qualities should be protected and that deviation outside of pre-established quality control conditions is a threat to ecological integrity. This could mean that either the structure/function of the system is at risk or has been damaged. Therefore, not only should one determine what attributes will be measured as indicators of ecological condition, but also one must state the acceptable bounds of variability for these measurements within which quality control conditions are being satisfied and those excursions beyond these limits that are unacceptable to the environmental management group.

Regardless of specific focus, ecological risk assessment strategies should be designed to estimate the probability of harm to at least three important attributes of natural systems:

1. **Self-maintenance or self-sustainability.**
   This means that the world's ecosystems have sufficient integrity to keep quality conditions within an acceptable range through time through natural processes. The caveat might be added that self-maintenance may be possible, even though the condition of the ecosystem is not ideal (defined as "the absence of any human intervention"). As a consequence, the system may not be pristine (as the word might be used by some theoretical ecologists), but still may be capable of self-maintenance.

2. **Protection of ecological capital.**
   From an anthropocentric view, society is interested in obtaining ecosystem services in a framework of sustained use. Examples of loss of ecological capital are summarized in Wilson (1988) and Ehrlich and Ehrlich (1990). Wilson's book represents a symposium jointly sponsored by the National Academy of Sciences and the Smithsonian Institute and focuses on the ecological capital represented by global biodiversity. That is, the global array of species, as yet incompletely inventoried, appears to be disappearing at an unprecedented rate (at least in historic times). This genetic information accumulated over millennia or longer is being lost at a much greater rate than it is being replaced. Other examples of ecological capital are topsoil, forests, and fossil water. It is a *sine qua non* that sustained use is not possible if ecological capital is being destroyed. Successional and other changes have been well-recognized for decades and depend on more appropriate genetic information's being available (packaged in species) as climatic and other conditions change. Thus, the structure and the function of an ecosystem and, consequently, the services it performs, are unimpaired, despite species replacement and other changes, if ecological integrity has been protected.

3. **Protection of commercially-, recreationally-, or aesthetically-valued species.**
   In some cases, such as the whooping crane, the Great Lakes Sports Fisheries, or the Oceanic Commercial Fishery, the interest is in protecting particular species from harm. At times, achieving this goal may lead to management practices not entirely compatible with the strategy of allowing ecosystems to undergo natural change.

In a very real sense, then, all ecological risk assessment is based on the probability of harm to ecological integrity, defined by Cairns (1977) as the maintenance of both structural and functional characteristics of the system at risk. From a societal standpoint, the most persuasive reason for doing so is to ensure that the ecosystem services from which humans benefit will be available on a sustained basis. In short, enlightened societies would wish future generations to enjoy at least the same benefits, and possibly enhanced benefits, compared to those presently available. Ecological risk assessment viewed in these terms is the probability of harm to the integrity of natural systems, or sometimes components thereof (particular species), as a result of toxic chemicals or other anthropogenic stress. In some cases, one might even include naturally-induced stress, such as that elicited by the Mt. St. Helens volcanic explosion (Franklin et al., 1988).

The use of ecosystem integrity as a focal point of ecological risk assessment has been criticized on the ground that this concept is too vague to be subject to formal quantitative analysis (e.g., Suter, 1990). While ecosystem integrity is not in itself a measurable end point, we argue here that this concept encompasses many important attributes that can be subjected to quantification. .• primary objective of this discussion is to provide a rationale for selecting appropriate, measurable indicators of ecosystem integrity. It is our contention that ecosystem integrity should represent a primary goal of ecological risk assessment, not that it is considered an end point by itself.

If one accepts those management goals for ecological risk assessment stated in previous paragraphs, then it is clear that much of the current framework for assessing ecological risk is inadequate for accomplishing stated objectives. Historically, ecological risk assessments have been reductionistic in their approach, restricted largely to evaluating the potential effect of a particular chemical on one or a few parameters of interest, generally commercially- or recreationally-valuable populations (e.g., a species of fish). Increases in the breadth and detail of ecological understanding argue for an expansion in the scope of ecological risk assessment. The approaches of Hunsaker and colleagues (1990), Graham et al. (1991), and others exemplify progress in this area. In this discussion, limitations are presented to current risk assessment protocols and an emerging strategy is outlined for expanding the decision-making process by including indicators of ecological integrity that can be used to monitor the status of important environmental attributes.

THE NEED FOR A MORE HOLISTIC APPROACH TO ECOLOGICAL RISK ASSESSMENT

Two general approaches have been used to evaluate the ecological risk associated with various human activities (Norton et al., 1988). A bottom-up strategy emphasizes laboratory experimentation and modelling to provide information with which to predict the fate and effects of different anthropogenic stressors that may be introduced into the environment. Ideally, such protocols are arranged in a tiered fashion (Kimerle et al., 1978), proceeding from relatively simple and inexpensive laboratory screening or rangefinding tests to predictive and confirmative tests (which are conducted both in controlled laboratory and field test systems) and, finally, the surveillance of natural receiving systems to validate the degree of risk estimated in experimental tiers of testing. Each tier culminates in a decision-making process, which is used to determine the amount and nature of testing required at successively higher, and inevitably more costly, tiers.

Literature on biological monitoring (e.g., Hellawell, 1978; Cairns et al., 1982; Morgan et al., 1986; Cairns, 1989) illustrates various aspects of a top-down approach to ecological risk assessment. As defined by Hellawell (1978), environmental monitoring involves an ongoing program of field surveys undertaken to ensure that previously-determined quality control standards are achieved and maintained. This error control capability must include a strategy for determining the cause of unacceptable environmental conditions and implementing remedial action. Ideally, corrective action should be taken before important ecosystem attributes and services are impaired; this requires a mechanism for predicting impending environmental impact with a reasonable degree of certainty so that pre-emptive action can be taken in a timely fashion. Much of the remainder of this discussion focuses on these two aspects of risk assessment.

Ecological risk assessment strategies largely have embraced a bottom-up approach for predicting the environmental impact of human activity on natural ecosystems and the services they provide. The need to test for the environmental effects of chemicals or other potential hazards in the laboratory prior to release into the environment is obvious. However, the inordinate reliance currently placed on this form of testing when assessing ecological risk is curious and requires additional explanation. Current risk assessment protocols aimed at conferring environmental protection belie their genesis in the form of mammalian toxicity tests and, later, as a response to concern over the cause of impacts on highly visible aquatic species, such as fish, that would predictably engender widespread public concern (e.g., Hart et al., 1945). While the focus of environmental protection efforts has expanded progressively to include protection of entire ecosystems, as well as selected attributes (e.g., target species), the strategy of most standardized risk assessment protocols essentially has remained unchanged. The use of arbitrary and, in some cases, conservative application factors in conjunction with single species test results undoubtedly has provided an adequate degree of protection to certain attributes of some ecosystems. However, the actual degree of environmental protection conferred by these procedures largely is unknown, simply because laboratory results rarely are validated in the natural ecosystems they are designed to protect (Cairns, 1983).

The degree to which laboratory tests alone are capable of predicting effects of anthropogenic stressors on natural ecosystems has been questioned on several grounds (e.g., NRC, 1981; Cairns, 1983 and 1986a; Ryder and Edwards, 1985; Kimball and Levins, 1985), including limitations in the extent to which: (1)effects documented in laboratory test systems, which are generally rather low in environmental realism, can be used to predict accurately the responses in the natural environment; (2)responses of surrogate species (i.e., standard laboratory test species) can be used to predict the responses of other species indigenous to a particular ecosystem; (3)response thresholds measured at one level of biological organization (e.g., population) can be used to predict effects at other levels (e.g., ecosystems); (4)general reliance on protocols that consider effects of different environmental stressors separately can predict potential environmental impacts that inevitably are cumulative; (5)the risk associated with all potential combinations of the thousands of chemicals in common use today possibly can be estimated.

There is substantial empirical evidence to support many of these contentions. The prevalence of synergistic and antagonistic interactions between co-occurring stressors (chemical interactions in particular) is well-known, but by no means thoroughly understood. The actual extent of exposure of biological material to a chemical may be magnified or mitigated within ecological food webs, compared to that measured in the environment (e.g., the water column or sediment in the case of aquatic ecosystems). There is compelling evidence (e.g., Mayer and Ellersieck, 1986) that individual species vary greatly in their relative sensitivity to different types of stressors. Results such as these falsify the assumption that a few standard "most sensitive" species ever can be identified and used as a basis for environmental protection (Cairns, 1986a), or that toxicity data from surrogate laboratory test species predict the susceptibility of indigenous species deemed to be of socioeconomic or ecological importance within a given ecosystem. Similarly, a scientifically-valid method for extrapolating responses observed at one level of biological organization (e.g., that of the individual or population) to predict impacts at other levels (e.g., communities and ecosystems) with a known degree of certainty has not been developed yet. Although the sensitivity of single species tests has been shown to be quite similar to that of experimental laboratory communities and ecosystems in many cases (e.g., Sloof et al., 1986), these same analyses also indicate that serious (e.g., greater than an order of magnitude) discrepancies in predictions among these two types of tests occur periodically (e.g., at least in five to ten percent of the chemicals tested). Even in cases where different levels exhibit similar sensitivities to stress, it is impossible for single species test data to predict the type of impacts that might be expected at higher levels, simply because community and ecosystem dynamics reflect interactive, as well as additive, effects of multiple responses at lower levels (see Cairns and Niederlehner, 1987). For example, while functional redundancy among component species may provide a buffer against change in ecosystem processes (e.g., primary productivity) in the face of environmental stress (Hill and Weigert, 1980; Schindler, 1987), loss of key "regulator species," such as keystone predators, can be magnified via direct and indirect population interactions to produce major shifts in membership and dominance within a community (e.g., Paine, 1966; Kerfoot and Sih, 1987; Carpenter, 1988).

Attempts have been made to resolve certain inadequacies in bottom-up risk assessment procedures. For example, certain chemical analyses (e.g., octanol-water partitioning coefficients, structure-activity relationships) have been shown to be reasonably good predictors of the potential for certain types of effects at different levels of biological organization (e.g., bioaccumulation in individuals and biomagnification in community food webs). Microcosm and mesocosm tests (Giesy, 1980; Hammons, 1981; Odum, 1984; Cairns, 1985 and 1986b; LaPoint et al., 1989) show increasing promise as standardized tools for providing more environmentally realistic assessments of higher level biological effects, those elicited by communities and ecosystems. Refinements such as these undoubtedly will confer environmental and economic benefits in the form of a reduction in unexpected environmental impact, costly remediation efforts, and unnecessary expenditures on pollution controls that produce no demonstrable benefits. However, continued improvements in laboratory-based test protocols still do not obviate the need for information on ecological risk and impact obtained from natural ecosystems.

DEVELOPING INDICATORS OF ECOLOGICAL INTEGRITY: INCORPORATING TOP-DOWN APPROACHES INTO ECOLOGICAL RISK ASSESSMENT

Top-down environmental assessments serve two principal functions as part of a comprehensive risk assessment strategy: (1)a periodic evaluation of environmental conditions (e.g., Hunsaker and Carpenter, 1990), and (2)an error detection and control device for ensuring that previously-stated environmental goals are maintained (Hellawell, 1978; Cairns et al., 1982). A monitoring strategy developed for these purposes should augment, not replace, existing laboratory-based protocols. Neither should these efforts be expected to supplant efforts to develop increasingly complex regional models for predicting risk (e.g., Hunsaker et al., 1990). Indeed, interfacing between these two strategies is crucial to the success of a risk assessment program.

Selecting Indicators of Ecosystem Integrity

The success of top-down assessments hinges on the selection of appropriate ecological indicators for gauging ecosystem integrity or health. The obvious dilemma is that everything is an indicator of something, but nothing is an indicator of everything. As a practical matter, economic and ecological considerations ensure that the number of indicators or attributes measured will be only a small fraction of those that conceivably might be measured. One might add the caveat that, even if more money were available for these measurements, the number of personnel capable of making them still would limit the actual number of measurements possible. Therefore, the selection of the attributes or indicators should not be on the basis of popularity among theoretical scientists, but rather on their value in making a decision on the probability of harm or risk to a particular ecosystem from a particular course of action or exposure. In short, the purpose of a strategy for information gathering should be to organize the process of data collection and analysis so that the information is synthesized in a systematic and orderly fashion that is cost-effective and, most importantly, best supports decision-making needs.

Literally thousands of useful ecological indicators have been proposed for various purposes related to ecological risk and impact assessment. No indicator is ideal for all purposes. Economic considerations, as well as the need to make management decisions in a timely manner, dictate that only a few of all possible parameters possessing indicator value will be measured routinely. Straightforward, objective guidelines are essential for selecting the most appropriate indicators for a particular purpose. Toward this end, various researchers have proposed desirable characteristics for indicators of ecological integrity (e.g., Ryder and Edwards, 1985; Kelly and Harwell, 1988; Suter, 1989; Hunsaker and Carpenter, 1990; Kerr, 1990). Several characteristics enhance the general utility of an ecological indicator, including that it (is):

1. Biologically relevant, that is, important in maintaining the normal appearance of the indigenous ecological community and unimpaired ecosystem operation;

2. Socially relevant, that is, the extent to which it provides information of obvious value and is easily related to relevant parties, including the lay public and politicians;

3. Sensitive to anthropogenic stressors;

4. Has low natural variability, which may confound responses to anthropogenic stress;

5. Broadly applicable to many stressors;

6. Diagnostic of the effects of a particular stressor;

7. Easily measured, that is, capable of being operationalized and quantified using standard methodologies with a known degree of performance and precision;

8. Interpretable to the extent that acceptable and unacceptable conditions can be distinguished readily in both a scientifically and legally defensible manner;

9. Cost-effective, maximizing the amount of relevant information gained per unit effort;

10. Minimally redundant in information content with other measured indicators;

11. Integrative, in that the parameter synthesizes information from, and therefore obviates the need for, measurement of many other potential indicators;

12. Has a historic database through which to define nominative variability, including cyclical and successional trends, such that a justifiable distinction between acceptable and unacceptable conditions can be drawn;

13. Maintains continuity in measurement over space, time, and a wide degree of environmental impact;

14. Relevant at appropriate scales of concern, that is, is concern focused on impacts at a local, regional and/or global scale?

15. Timely, that is, the rapidity with which raw measurements can be converted into effective management actions;

16. Anticipatory of impending environmental degradation, that is, capable of providing an early warning signal leading to pre-emptive action;

17. Nondestructive to the specific indicator and ecosystem(s) of concern.

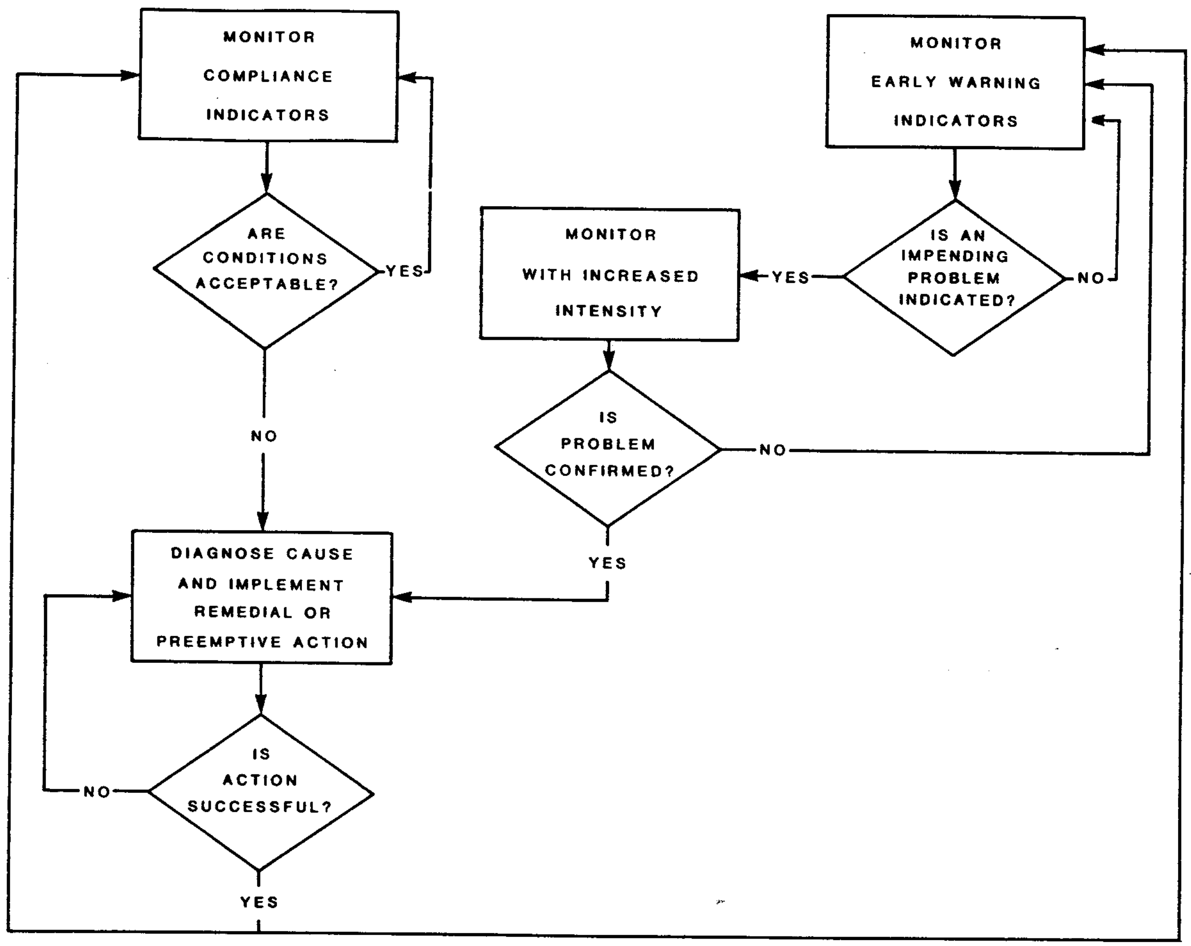
Certain characteristics are desirable for any indicator, no matter what the specified purpose. Ideally, any indicator should be sensitive to environmental stress and easily measurable at scales appropriate to the management problem being addressed. Given the expense of any monitoring program, cost effectiveness always is an important criterion. This includes the minimization of redundancy in information content among measured indicators. Availability of a historical database provides important baseline information, especially in ecosystems or regions where reference (i.e., minimally-affected) sites currently are not available due to extensive impact. Certain criteria obviously are mutually exclusive; for example, an indicator that is responsive to a wide range of anthropogenic impacts (criterion #5) cannot be diagnostic of a particular stressor (criterion #6). In fact, the importance of a particular characteristic will vary, depending on the type of indicator considered.

Indicator selection should be carried out within a framework that addresses three critical questions related to risk assessment and ecosystem management: (1)Are stated objectives being met? (2)If stated objectives are not being met, what is the cause of such failure? (3)Can pending failures be predicted before unacceptable impact actually occurs? To answer these questions, a quality control monitoring program must fulfill multiple purposes. The first and most obvious purpose is to provide an ongoing assessment of environmental conditions to determine whether quality control goals and objectives are being achieved. A second purpose of monitoring is to identify corrective actions in the event that management objectives (i.e., quality control conditions) are not being met. Demonstrating the cause of environmental degradation is a much more difficult task than merely observing that deleterious effects have occurred. No single diagnostic method is likely to be suitable in all situations. However, certain guidelines can be useful. Gilbertson (1984) proposes a three-step diagnostic process: (1)identification of the environmental impact; (2)epidemiology, the process of determining the extent and nature of these effects, and the formulation of causal hypotheses; (3)etiology, which involves experimentation with the suspected environmental stressor and other stressors known to exhibit similar effects, in order to reach conclusions regarding causation. These conclusions must be sufficiently robust to allow for rehabilitation strategies to be implemented to correct the problem if damage has occurred. Preferably, identification and diagnosis of a problem should occur sufficiently early so that corrective actions can be taken before substantive damage has occurred.

It is extremely improbable that any single indicator can fulfill all the above-stated needs. Furthermore, since no indicator is completely reliable, whether due to occasional spurious responses or measurement errors, multiple lines of confirming evidence (that is, either corresponding and, therefore, strengthening the conclusion or validating a prediction and, therefore, indicating the model is robust) are extremely helpful when making decisions regarding current environmental conditions and future management actions. In order to further a comprehensive and organized approach to ecological risk assessment and management, three general types of indicators are recommended for development (Figure 1): (1)compliance indicators, (2)diagnostic indicators, and (3)early warning indicators. Desirable characteristics for each of these are summarized on the following pages and in Table 1.

Figure 1. A Top-Down Approach to Ecological Risk Assessment Utilizing Three Classes of Indicators

Compliance indicators are chosen for monitoring that provide suitable end points for judging the acceptability of environmental conditions. In the event that conditions are judged to be unacceptable, diagnostic indicators are used to isolate causative agents and suggest remedial action. Early warning indicators are chosen to complement compliance indicators by signalling impending deterioration in environmental conditions, so that preventative measures can be taken to avoid the onset of unacceptable conditions. To minimize the chance that a false positive signal will result in unnecessary action, positive signals from one indicator should be confirmed in a timely manner by intensifying monitoring activity to include other suitable indicators.



Compliance Indicators

Compliance indicators are those chosen to confirm that previously established environmental quality conditions are being met. For years, environmental professionals have been accustomed to generating compliance indicators for chemical-specific objectives for both air and water quality. While there is dissension among environmental professionals as to the specific biological indicators that are most useful for judging compliance with broad ecosystem objectives (e.g., maintaining ecosystem integrity), there is widespread agreement regarding the general characteristics that such measures should possess. Given that the number of indicators that reasonably can be measured is limited, the most effective indicators of compliance with environmental quality control objectives are those that integrate many characteristics related to the stated objective. The concept of the "integrator organism" is discussed at length by Ryder and Edwards (1985). However, there is justification for considering parameters or attributes at other levels of biological organization, such as the community and ecosystem, for this purpose. Thus, one would know that compliance had been achieved at more than one level of biological organization. Difficulties in extrapolating responses among species and levels of biological organization discussed earlier for laboratory testing apply here as well.

Table 1. Desirable Attributes of Indicators Used for Different Purposes

Importance of a particular attribute is designated using a scale from 1 (less important) to 3 (essential). Certain characteristics are universally desirable and are marked with an asterisk.

Indicator Characteristic	Type of Indicator: Compliance	Diagnostic	Early Warning
Biological Relevance	3	2	2
Social Relevance	3	2	2
Sensitivity	*	*	*
Low Natural Variability	*	*	*
Broad Applicability	2	1	2
Diagnostic Capability	1	3	1
Easily Measured	*	*	*
Interpretable	3	1	2
Cost-effective	*	*	*
Minimum Redundancy	2	1	1
Integrative	2	1	1
Historical Database	*	*	*
Continuity	3	1	1
Appropriate Scale	*	*	*
Timeliness	2	3	3
Anticipatory	1	1	3
Nondestructive	*	*	*

Compliance indicators should be the most obvious part of any environmental monitoring effort, and, thus, their significance should be communicated readily to both citizens and policymakers in a manner that facilitates decisionmaking. Individual or population attributes of commercially-/ aesthetically-important species (e.g., coho salmon or bald eagles), for example, are useful as compliance indicators for this and other reasons. To give a parallel example, a compliance indicator of economic condition might be the gross output of goods and services for the region. Since society and the economy depend on both economic and ecological well-being, compliance indicators in both categories are essential. To the extent that long-term trends must be evaluated, as generally is the case for efforts at restoration or rehabilitation, continuity of measured indicators also is crucial.

Gauging fulfillment of broader goals of environmental protection, such as those for ecological integrity proposed here, will require that ecologically-important parameters and processes be monitored as well. Changes in the behavior and population dynamics of keystone species, those which fulfill functional roles upon which many other species depend (e.g., top predators or pollinators), may be useful for this purpose. Other examples of suitable indicators of ecological integrity might include: (1)shifts in species composition/functional redundancy within the indigenous community; (2) changes in ecologically-important processes (e.g., primary productivity, decomposition, nutrient spiralling); and (3)changes in the coverage of ecologically important habitats (e.g., wetlands or spawning beds).

Diagnostic Indicators

By definition, a monitoring program must include a mechanism for determining the cause of noncompliance and suggesting corrective action. One should not assume automatically that those attributes most useful for judging compliance with a specific objective also are best for determining why objectives have not been met. Causes of environmental deterioration are not always obvious or simple and, thus, may not be determined easily without a systematic and orderly protocol for data collection and review.

Diagnostic indicators are selected for their ability to isolate causative agents. Given this charge, diagnostic tools are, by necessity, reductionistic in their focus and, thus, unlike compliance indicators, may be applied only to specific sites/ conditions (e.g., controlled field or laboratory assessments). Once an unacceptable condition is identified, a timely diagnosis of the cause may reduce greatly the extent of environmental damage. While biological and social relevance are advantageous, they are not as essential as they are for compliance indicators.

The number of probable causative agents can be narrowed by correlating deleterious changes in parameters of interest with shifts in other environmental variables; obviously, all measures considered relevant to a particular situation (e.g., toxic substances known to be discharged into the ecosystem under study) must be included in the monitoring program for such an approach to be implemented successfully. Information on changes in the quantity or quality of ecologically-important habitats or resources or the water column concentration of a toxic chemical, for example, may be associated with changes in performance of biological indicators (e.g., changes in coho salmon population or aquatic food web dynamics). Establishment of these types of relationships is an important part of monitoring programs, such as the Environmental Monitoring and Assessment Program (EMAP) currently under development at the Environmental Protection Agency (Hunsaker and Carpenter, 1990). Correlative relationships are useful for hypothesis generation about potential causes, but alone they do not provide strong evidence for cause-effect linkages (e.g., those which would warrant initiation of expensive and costly corrective actions). The category of diagnostic indicators proposed herein includes specific changes (e.g., enzyme changes induced by bioaccumulation of substitute toxics levels) that are capable of isolating specific stressor effects on compliance indicators.

Not all diagnostic information need be gathered *in situ*, and, indeed, one of the strongest interfaces between top-down and bottom-up assessments should occur at the stage of diagnosis. Data gathered from controlled laboratory and mesocosm tests often can provide critical information for isolating the causative agent eliciting a particular effect on important ecosystem parameters. Chemical fractionation of ambient water followed by toxicity testing often provides useful diagnostic information (e.g., Mount and Anderson-Carnahan, 1988).

Diagnostic information tends to be extremely stressor- and site-specific, such that no single diagnostic protocol can be applied to all situations. In many instances, both laboratory and field information must be gathered in order to determine the cause of the impact and to suggest remedial action.

Early Warning Indicators

Since the object of any ecological risk assessment is to protect ecosystem integrity, an early warning of impending harm is preferable to remedial action later. Thus, parameters and processes selected to provide an early warning capability in natural ecosystems must, above all, anticipate impending environmental damage before it occurs. Early warning signals also must be capable of being processed and interpreted in a timely fashion if they are to fulfill their designated purpose of avoiding unacceptable and costly environmental damage. As with most compliance indicators, early warning indicators are part of an ongoing surveillance program, thus making continuity in time (and space) an essential feature. As with diagnostic indicators, biological and social relevance is a desirable, but not essential, feature of early warning indicators.

In contrast to the success of on-line monitoring systems using sensitive physiological parameters (e.g., fish ventilation rate) to signal the onset of undesirable conditions resulting from point source effluents (e.g., Cairns and Gruber, 1980), identification of suitable early warning indicators of environmental stress amenable to field assessment has been an extremely difficult task. The early warning capability of monitoring programs relying solely on traditional indicators of environmental condition (e.g., physico-chemical conditions or indicator species) generally is quite low; by the time deleterious changes in water quality, individual health, or population dynamics are observed, substantial environmental deterioration often has occurred already. Use of species with short generation times (e.g., prokaryotic and eukaryotic microbes) minimizes this lag period between stress induction and stress detection (Cairns and Pratt, 1989). Measurements of ecosystem processes have been proposed as potential early warning indicators of stress in both aquatic and terrestrial systems (Van Voris et al., 1980; Bormann, 1983; Odum, 1985; Rapport et al., 1985). Sensitive processes may include rates of decomposition, primary productivity, and nutrient spiralling. However, the reliability of such measures is questionable, given the degree of natural temporal and spatial variation and the low precision of measurement in many cases. For these and other reasons, structural parameters have been advocated over functional measures for development as early warning indicators (e.g., Schindler, 1987).

Measurements at lower levels of biological organization may provide useful early warning indications of changes in some important ecosystem attributes (e.g., populations of commercially- or ecologically-important species). One area of research that promises to enhance the early warning capability of environmental monitoring programs is that on "biomarkers," measures of genetic, immunological, and enzymatic activity that indicate stress in individual organisms (see Digiulio, 1989; McCarthy and Shugart, 1990). While stress-specific biomarkers may provide powerful diagnostic tools for determining the cause of environmental deterioration, generic responses (e.g., rate-limiting enzymes in key biochemical pathways) offer the best prospect for development as broadly applicable early warning signals. Prospects for the use of biomarkers as early warning indicators of environmental stress are discussed in greater detail by Giesy et al. (1988).

PREDICTING ECOSYSTEM RISK IN THE FACE OF UNCERTAINTY

Any effort to assess the harm that various human activities pose to ecosystem integrity entails a certain degree of uncertainty. Natural ecosystems are complex entities, and many aspects of their operation remain poorly understood, despite acknowledged increases in the extent of ecological understanding. Furthermore, there is a paucity of information regarding natural variability in many ecosystem attributes potentially useful for risk assessment. Inaccuracies or imprecisions in available sampling methodologies can contribute substantial error to the assessment, depending on the particular measurements involved. These limitations are commonplace in ecological research and are exacerbated when ecological information is used to assess the risk posed by human activities, many of which elicit novel and poorly understood responses from natural ecosystems. Consequently, it must be expected that management decisions sometimes will result in unanticipated changes in important ecosystem attributes, some of which will be detrimental.

Although the sources of uncertainty vary depending on the risk assessment strategy in question, most involve extrapolations based on limited knowledge. As already discussed, bottom-up approaches suffer from uncertainty generated by extrapolating from simplistic laboratory situations to complex natural ecosystems. Despite attempts to quantify the uncertainty associated with these procedures (e.g., Suter et al., 1985; Sloof et al., 1986), there remains no scientifically justifiable means for extrapolating between levels of biological complexity (i.e., from the individual to the community). While laboratory-based approaches adequately may predict effects associated with localized environmental problems, such as a point source discharge of a toxic chemical, they are incapable of predicting reliably effects caused by environmental changes occurring over broader spatial scales and heterogeneous landscapes, such as the effects of acid deposition or regional and global climate change.

Ecosystem-based approaches provide a capability with which to address and reduce directly the uncertainty associated with traditional risk assessment procedures. The successful implementation of this type of approach is hampered principally by limitations in ecological understanding. While the importance of ecosystem complexity and heterogeneity is recognized widely, the most appropriate means of addressing these issues is far from resolved. As discussed by May (1989), a predictive understanding of the ecological consequences of a particular management option requires information at several levels of biological organization. This problem is not merely one of extrapolating from lower to higher levels of biological organization. For example, an understanding of energy and materials fluxes between ecosystem compartments may yield an unreliable prediction of the importance of a species in regulating community structure, since many keystone species (e.g., pollinators) account for a disproportionately small amount of energy flow. A predictive understanding of regional environmental problems requires an understanding of spatial heterogeneity and its influence on ecosystem responses (Hunsaker et al., 1990; Holland et al., 1991).

Simulation models have been embraced as one means of improving quantitative risk assessment (e.g., Graham et al., 1991). Computer models are extremely useful as a means of generating hypotheses about how ecosystems operate and respond to anthropogenic perturbations. However, the predictive capability of these techniques is limited strictly by the availability and reliability of empirical data. In order to remain tractable, most computer models require that the scope and complexity of the environmental problem be reduced; thus, like traditional risk assessment techniques, it is questionable whether this approach is suitable for addressing complex problems involving multiple impacts and responses. Other types of uncertainty associated with model predictions increase with model complexity, as more assumptions regarding real world processes are required. Thus, despite the heuristic value of mathematical models, there currently is little precedent for relying on this approach for generating predictions in the context of ecological risk assessment. At the very least, model predictions must be validated in real ecosystems.

The predictive capability of the ecosystem-based approach described here is contingent upon the selection of suitable early warning indicators of ecosystem stress. Development of an early warning detection system must include an evaluation of: (1)the reliability with which truly significant changes in indicator status can be discerned from normal background variation and (2)the extent to which a significant change in indicator status accurately predicts deleterious changes in ecosystem attributes deemed important. Given that no indicator will be completely reliable in either of these regards, it is necessary to determine the likelihood for the generation of false signals by the early warning detection system. A false negative would be an indication that there was no risk to the corresponding ecosystem attribute when, in fact, unacceptable environmental damage was occurring. A false positive, of course, would be a signal that ecological integrity was being impaired when, in fact, this was not occurring. Quantification of the uncertainty associated with individual early warning indicators would allow for management actions to be gauged to both the number and reliability of the signals received. Numerous reliable signals would demand immediate corrective action, whereas fewer or less reliable signals merely may warrant a closer inspection of ecosystem condition.

CONCLUSION

No risk assessment strategy, no matter how comprehensive or costly, realistically can be designed so as to be infallible. Two efforts, the Pellston Series of hazard evaluation workshops (Cairns et al., 1978) and the National Research Council Committee on Determining the Effects of Chemicals on Ecosystems (NRC, 1981), illustrate the extent to which uncertainty and surprise will continue to be ingrained in risk assessment procedures for the foreseeable future. In both cases, attempts were made to use information existing at the time to determine whether, with hindsight, it was possible to predict phenomena that had caused previous major pollution problems. The conclusions were equivocal. For example, while the methyl mercury problem could have been predicted rather easily by the inclusion of simple sediment-water microcosms into test protocols, it is unlikely that eggshell thinning in birds resulting from DDT could have been predicted using any available test procedures or models. Certainly, several other examples can be found to support conclusions on both sides. While properly designed risk assessment strategies can minimize the number of "surprises" encountered by ecosystem managers, conceptual and methodological limitations preclude the ability to make accurate predictions in all cases. Thus, while predictive approaches to risk assessment obviously are crucial, reactive control systems also will continue to be essential.

The inclusion of properly designed ecosystem monitoring programs into ecological risk assessments can contribute greatly to a reduction in uncertainty and surprise. In particular, judicious selection of indicators of ecological integrity can provide a framework for maintaining ecosystem attributes and services deemed important by society. Inadequacies in current understanding of the factors contributing to ecosystem operation and integrity, as well as economic and political constraints, inevitably will limit the overall effectiveness of such a program. While the suitability of any program of objectives and indicators may be less than ideal, surely an incomplete assessment of environmental conditions will yield more benefits than none at all.

ACKNOWLEDGMENTS

The authors are indebted to Darla Donald for editorial assistance in the preparation of this manuscript and to B.R. Niederlehner for developing Table 1.

REFERENCES

Bormann, F.H. 1983. Factors Confounding the Evaluation of Air Pollutant Stress on Forests: Pollutant Input and Ecosystem Complexity. In *Symposium, Acid Deposition, a Challenge for Europe*. Karlsruhe, Federal Republic of Germany, pp. 19-23.

Cairns, J., Jr. 1977. Quantification of Biological Integrity. In *Integrity of Water*, R.K. Ballentine and L.J. Guarria, eds. U.S. Environmental Protection Agency, Office of Water and Hazardous Materials, Washington, DC, pp. 171-187.

Cairns, J., Jr. 1983. Are Single Species Tests Alone Adequate for Estimating Environmental Hazard? *Hydrobiologia* 100: 47-57.

Cairns, J., Jr., ed. 1985. *Multispecies Toxicity Testing*. Pergamon Press, New York.

Cairns, J., Jr. 1986a. The Case for Direct Measurement of Environmental Responses to Hazardous Materials. *Water Resources Bulletin* 22: 841-842.

Cairns, J., Jr., ed. 1986b. *Community Toxicity Testing*, STP 920. American Society for Testing and Materials, Philadelphia, PA.

Cairns, J., Jr. 1989. Biological Montioring of Water Quality. In *International Symposium on the Design of Water Quality Information Systems*, R.C. Ward, J.C. Loftis, and G.B. McBride, eds. Fort Collins, CO, pp. 338-346.

Cairns, J., Jr., and D. Gruber. 1980. A Comparison of Methods and Instrumentation of Biological Early Warning Systems. *Water Resources Bulletin* 16: 261-266.

Cairns, J., Jr., and B.R. Niederlehner. 1987. Problems with Selecting the Most Sensitive Species for Toxicity Testing. *Hydrobiologia* 153: 87-94.

Cairns, J., Jr., and J.R. Pratt. 1989. Advantages and Disadvantages of Employing Microorganisms as Indicators of Pollution. *Current Practices in Environmental Science and Engineering* 4: 111-123.

Cairns, J., Jr., K.L. Dickson, and A. Maki, eds. 1978. *Estimating the Hazard of Chemical Substances to Aquatic Life*. American Society for Testing and Materials, Philadelphia, PA.

Cairns, J., Jr., A.L. Buikema, Jr., D.S. Cherry, E.E. Herricks, W.H. van der Schalie, R. Matthews, B.R. Niederlehner, and J.H. Rodgers, Jr. 1982. *Biological Monitoring*. Pergamon Press, London.

Carpenter, S.R., ed. 1988. *Complex Interactions in Lake Communities*. Springer-Verlag, New York.

Digiulio, R.T. 1989. Biomarkers. In Ecological Assessment of Hazardous Waste Sites: A Field and Laboratory Reference. EPA/600/3-89/013. United States Environmental Protection Agency, Cincinnati, OH, pp. 7-1 to 7-34.

Ehrlich, P.R., and A.H. Ehrlich. 1990. *The Population Explosion*. Simon and Schuster, New York.

Franklin, J.F., P.M. Frenzen, and F.J. Swanson. 1988. Re-Creation of Ecosystems at Mount St. Helens: Contrasts in Artificial and Natural Approaches. In *Rehabilitating Damaged Ecosystems*, Vol. II, J. Cairns, Jr., ed. CRC Press, Boca Raton, FL, pp. 1-38.

Giesy, J.P., ed. 1980. *Microcosms in Ecological Research*. National Technical Information Center, Springfield, VA.

Giesy, J.P., D. Versteeg, and R. Graney. 1988. A Review of Selected Clinical Indicators of Stress-Induced Changes in Aquatic Organisms. In *Toxic Contaminants and Ecosystem Health: A Great Lakes Focus*, M.S. Evans, ed. John Wiley and Sons, NY, pp. 169-200.

Gilbertson, M. 1984. Need for Development of Epidemiology for Chemically Induced Diseases in Fish in Canada. *Canadian Journal of Fisheries and Aquatic Sciences* 41: 1534-1540.

Graham, R.L., C.T. Hunsaker, R.V. O'Neill, and B.L. Jackson. 1991. Ecological Risk Assessment at the Regional Scale. *Ecological Applications* 1: 196-206.

Hammons, A., ed. 1981. *Methods for Ecological Toxicology*. Ann Arbor Science, Ann Arbor, MI.

Hart, W.B., P. Douderoff, and J. Greenbank. 1945. The Evaluation of the Toxicity of Industrial Wastes, Chemicals and Other Substances to Fresh Water Fishes. Waste Control Laboratory, Atlantic Refining Company, Philadelphia, PA.

Hellawell, J.M. 1978. *Biological Surveillance of Rivers*. Water Research Centre, Stevenage, England.

Hill, J., and R.G. Wiegert. 1980. Microcosms in Ecological Modeling. In *Microcosms in Ecological Research*, J.P. Giesy, Jr., ed. National Technical Information Center, Springfield, VA, pp. 138-163.

Holland, M.M., P.G. Risser, and R.J. Naiman, eds. 1991. *Ecotones: The Role of Landscape Boundaries in the Management and Restoration of Changing Environments*. Chapman and Hall, New York.

Hunsaker, C.T., and D.E. Carpenter, eds. 1990. *Environmental Monitoring and Assessment Program: Ecological Indicators*. Office of Research and Development, United States Environmental Protection Agency, Research Triangle Park, NC.

Hunsaker, C.T., R.L. Graham, G.W. Suter, R.V. O'Neill, L.W. Barnthouse, and R.H. Gardner. 1990. Assessing Ecological Risk on a Regional Scale. *Environmental Management* 14: 325-332.

Kelly, J.R., and M.A. Harwell. 1988. Indicators of Ecosystem Response and Recovery. In *Ecotoxicology: Problems and Approaches*, S.A. Levin, M.A. Harwell, J.R. Kelly, and K.D. Kimball, eds. Springer-Verlag, New York, pp. 9-40.

Kerfoot, W.C., and A. Sih, eds. 1987. *Predation: Direct and Indirect Impacts on Aquatic Communities*. University Press of New England, Hanover, NH.

Kerr, A. 1990. Canada's National Environmental Indicators Project: Background Report. Sustainable Development and State of the Environment Reporting Branch, Ottawa.

Kimball, K.D., and S.A. Levins. 1985. Limitations to Laboratory Bioassays: The Need for Ecosystem-Level Testing. *Bioscience* 35: 165-171.

Kimerle, R.A., W.E. Gledhill, and G.L. Levinskas. 1978. Environmental Safety Assessment of New Materials. In *Estimating the Hazard of Chemical Substances to Aquatic Life*, STP 657, J. Cairns, Jr., K.L. Dickson, and A.W. Maki, eds. American Society for Testing and Materials, Philadelphia, PA, pp. 132-146.

LaPoint, T.W., J. F. Fairchild, E.E. Little, and S.E. Finger. 1989. Laboratory and Field Techniques in Ecotoxicological Research:

Strengths and Limitations. In *Aquatic Ecotoxicology: Fundamental Concepts and Methodologies*, Vol. II, A. Boudou and F. Ribeyre, eds. CRC Press, Boca Raton, FL, pp. 239-255.

May, R.M. 1989. Levels of Organization in Ecology. In *Ecological Concepts: The Contribution of Ecology to an Understanding of the Natural World*, J.M. Cherrett, ed. Blackwell Scientific Publications, Oxford, pp. 339-363.

Mayer, F.L., Jr., and M.R. Ellersieck. 1986. Manual of Aquatic Toxicity: Interpretation and Data Base for 410 Chemicals and 66 Species of Freshwater Animals. Resource Publication 160, U.S. Department of Interior, Washington, DC.

McCarthy, J.F., and L.R. Shugart. 1990. *Biomarkers of Environmental Contamination*. Lewis Publishers, Inc., Boca Raton, FL.

Morgan, E.L., K.W. Eagleson, T.P. Weaver, and B.G. Isom. 1986. Linking Automated Biomonitoring to Remote Computer Platforms with Satellite Data Retrieval in Acidified Streams. In *Impact of Acid Rain and Deposition on Aquatic Biological Systems*, STP 928, B.G. Isom, S.D. Dennis, and J.M. Bates, eds. American Society for Testing and Materials, Philadelphia, PA, pp. 84-91.

Mount, D.I., and L. Anderson-Carnahan. 1988. Methods for Aquatic Toxicity Identification Evaluation. EPA/600/3-88-035. National Technical Information Service, Springfield, VA.

(NRC) National Research Council. 1981. *Testing for Effects of Chemicals on Ecosystems*. National Academy Press, Washington, DC.

Norton, S., M. McVey, J. Colt, J. Durda, and R. Hegner. 1988. Review of Ecological Risk Assessment Methods. EPA/230-10-88-041. National Technical Information Service, Springfield, VA.

Odum, E. P. 1984. The Mesocosm. *Bioscience* 34: 558-562.

Odum, E. P. 1985. Trends Expected in Stressed Ecosystems. *Bioscience* 35: 419-422.

Paine, R.T. 1966. Food Web Complexity and Species Diversity. *American Naturalist* 100: 65-75.

Rapport, D.J., H.A. Regier, and T.C. Hutchinson. 1985. Ecosystem Behavior Under Stress. *American Naturalist* 125: 617-640.

Ryder, R.A., and C.J. Edwards. 1985. A Conceptual Approach for the Application of Biological Indicators of Ecosystem Quality in the Great Lakes Basin. Report to the Great Lakes Science Advisory Board, Windsor, Canada.

Schindler, D.W. 1987. Detecting Ecosystem Responses to Anthropogenic Stress. *Canadian Journal of Fisheries and Aquatic Science* 44(Suppl. 1): 6-25.

Sloof, W., J.A.M. van Oers, and D. de Zwart. 1986. Margins of Uncertainty in Ecotoxicological Hazard Assessment. *Environmental Toxicology and Chemistry* 5: 841-852.

Suter, G.W. 1989. Ecological Endpoints. In Ecological Assessment of Hazardous Waste Sites: A Field and Laboratory Reference. EPA 600/3-89-013. W. Warren-Hicks, B. Parkhurst, and S. Baker, Jr., eds. National Technical Information Service, Springfield, VA.

Suter, G.W. 1990. Endpoints for Regional Ecological Risk Assessments. *Environmental Management* 14: 9-23.

Suter, G.W., L.W. Barnthouse, J.E. Breck, R.H. Gardner, and R.V. O'Neill. 1985. Extrapolating from the Laboratory to the Field: How Uncertain are You? In *Aquatic Toxicology and Hazard Assessment: Seventh Symposium*, ASTM STP 854, R.D. Cardwell, R. Purdy, and R. C. Bahner, eds. American Society for Testing and Materials, Philadelphia, PA, pp. 400-413.

Van Voris, P., R.V. O'Neill, W.R. Emanuel, and H.H. Shugart, Jr. 1980. Functional Complexity and Ecosystem Stability. *Ecology* 61: 1252-1360.

Wilson, E., ed. 1988. *Biodiversity*. National Academy Press, Washington, DC.