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Council of Great Lakes Research Managers

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A Proposed Framework for Developing

INDICATORS of ECOSYSTEM HEALTH

for the Great Lakes Region



International Joint Commission
United States and Canada

1991

Council of Great Lakes Research Managers
Report to the International Joint Commission

A Proposed Framework for Developing

INDICATORS
Of
ECOSYSTEM HEALTH

for the Great Lakes Region

July, 1991

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International Joint Commission
United States and Canada

DISCLAIMER

This report to the International Joint Commission was carried out as part of the activities of the Council of Great Lakes Research Managers. While the Commission supported this work, the specific conclusions and recommendations do not necessarily represent the views of the International Joint Commission, its Boards, Council or Committees.

ACKNOWLEDGEMENT

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The study team formed to guide the work included Al Davidson, Rick Burnett and Peter Seidl. The final report was edited at the International Joint Commission by Peter Seidl and Pat Murray, with graphics by Bruce Jamieson.

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Cover photo depicts purple loosestrife.

Preamble

The revised Great Lakes Water Quality Agreement as amended by Protocol, signed November 18, 1987, increased the responsibility of the Parties in terms of achieving progress in restoring and maintaining the chemical, physical and biological integrity of the waters of the Great Lakes Basin Ecosystem. The International Joint Commission, under Article VII, has the responsibility of evaluating progress toward achieving the goals of the Great Lakes Water Quality Agreement. Specifically, the Commission was charged with assessing the effectiveness of the programs carried out by the Parties.

Evaluating the progress of any program requires that benchmarks or standards be established to judge whether goals have been achieved. Because the ecosystem approach requires measuring the integrity of the ecosystem, it is no longer adequate to simply measure emissions, discharges and loadings alone. Instead, we must measure ecosystem components that indicate system integrity.

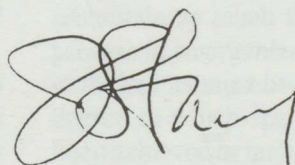
In 1990, the International Joint Commission established priorities. The Commission recognized that measuring ecosystem integrity was essential in carrying out its assessment responsibilities and named indicators of ecosystem health as a high priority. The indicators needed would tell the Commission whether ecosystem integrity was being restored and maintained.

Ecosystem indicators should serve a similar function in evaluating the state of the environment as economic indicators do in evaluating the financial health of a nation. Numerous economic indicators are needed to assess trends in the economy. No one indicator is able to serve the diverse needs of investment, money supply, business decisions, tax policy and numerous other considerations. Ecological indicators also serve different functions and more than one or combinations may be needed to adequately assess the state of the environment.

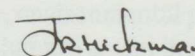
The ultimate test of indicators is whether they are useful to decision makers. Will the Commission be able to assess whether progress is adequate? Will indicators be measurable? Do the goals need to be formulated so that they are measurable along with indicators?

The Council of Great Lakes Research Managers commissioned a study of indicators. The objective was to define the uses and limitations of indicators and establish the characteristics needed for good indicators.

The Council agrees that this report is a comprehensive summary of what is known about indicators but is a cursory attempt to show the linkages between socio-economic and biogeochemical variables. This report has defined a valid starting point for us and will facilitate further discussions on indicators of ecosystem health.



Jon G. Stanley
United States CoChair



J. Roy Hickman
Canadian CoChair

1992-1993

The Council of British Columbia Indians is pleased to announce the results of the 1992-1993 election. The Council has elected a new President, Vice President, and a new Board of Directors. The Council also elected a new Executive Director. The Council will continue to work for the best interests of the people of British Columbia.

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J. R. Johnson
Executive Director


J. R. Johnson
Executive Director

1.

Introduction

As the complexity of human impact on the environment increases and our ecological capital shrinks, the need for effective management of our natural resources becomes increasingly critical. The nature of environmental impact has changed radically since the beginning of the Industrial Revolution, as exemplified by the increasing importance of regional (e.g. acid precipitation) and global (e.g. climate change) nonpoint sources of stress relative to point source discharges, the increasing number of potential stressors and the importance of cumulative impacts. In the Great Lakes region, as elsewhere, the focus of environmental protection has broadened from the development of stress-specific water quality standards to the achievement of broad objectives for restoring self-maintaining ecosystems and the maintenance of the quality of human life. To address these changes in the way which society affects and wishes to restore and protect the environment will require improvements in the effectiveness of environmental management strategies.

There are two approaches to the evaluation of environmental degradation at the community and ecosystem level (Norton 1988; Hunsaker and Carpenter, 1990). Directly assessing changes in communities and ecosystems in the natural environment, then subsequently diagnosing problems and causative agents is a "top-down" method. Literature on biological monitoring illustrates this approach (e.g. Hellawell 1978). In contrast, "bottom-up" methods use laboratory data showing effects on simple systems to model changes in the more complex natural ecosystem. Hazard assessment protocols illustrate this approach (Cairns et al. 1978). Routine bottom-up procedures for estimating hazard (e.g. laboratory testing with human or ecosystem surrogates, models of fate and transport) are limited in their ability to predict impacts on natural ecosystems for several reasons (National Research Council 1981; Cairns 1983; 1986; Ryder and Edwards, 1985; Kimball and Levins, 1985), including:

- Difficulties involved in the use of effects observed in the laboratory to predict responses in the natural environment
- Difficulties involved in using the response of relatively simple biological test systems (e.g. single-species laboratory bioassays) to predict effects on relatively complex systems (e.g. natural ecosystems)

- Use of protocols that consider the effect of each type of stress separately, even though impacts are inevitably cumulative in the environment
- Inability to test all possible combinations of the thousands of chemicals in common use today.

Natural ecosystems are complex, multivariate systems and are being simultaneously exposed to a multitude of stresses, the mechanisms and cumulative effects of which are poorly understood. Thus, it seems unlikely that the successful management of major ecosystems, such as the Laurentian Great Lakes, to achieve broad environmental and socio-economic objectives is possible without a substantial broadening of the environmental assessment framework to encompass top-down ecosystem objectives.

Periodic direct observation of the health of communities in their natural environment affords the opportunity to validate predictions of impact in the real world from bottom-up methods and provides mechanisms for integrating corrective actions into the management plan. This iterative process is described by the term "biological monitoring," the ongoing assessment of environmental conditions to insure that previously formulated objectives are being maintained (Figure 1; Hellawell 1978).

Every measurable parameter has some value with regard to assessing environmental conditions. However, because it is impossible to measure every environmental variable or to assimilate so much information into the decision making process in an organized manner, environmental parameters or indicators must be selected that are useful in judging the degree to which specified environmental conditions have been achieved or maintained. An indicator is "a characteristic of the environment that, when measured, quantifies the magnitude of stress, habitat characteristics, degree of exposure to the stressor, or degree of ecological response to the exposure" (Hunsaker and Carpenter, 1990). The number of potential indicators is infinite and selection of the few "best" indicators from this vast array is by no means a simple exercise. Indicator parameters serve several purposes in the context of environmental monitoring. Several disparate, and sometimes conflicting, considerations are involved in selecting the most appropriate indicators for a particular purpose.

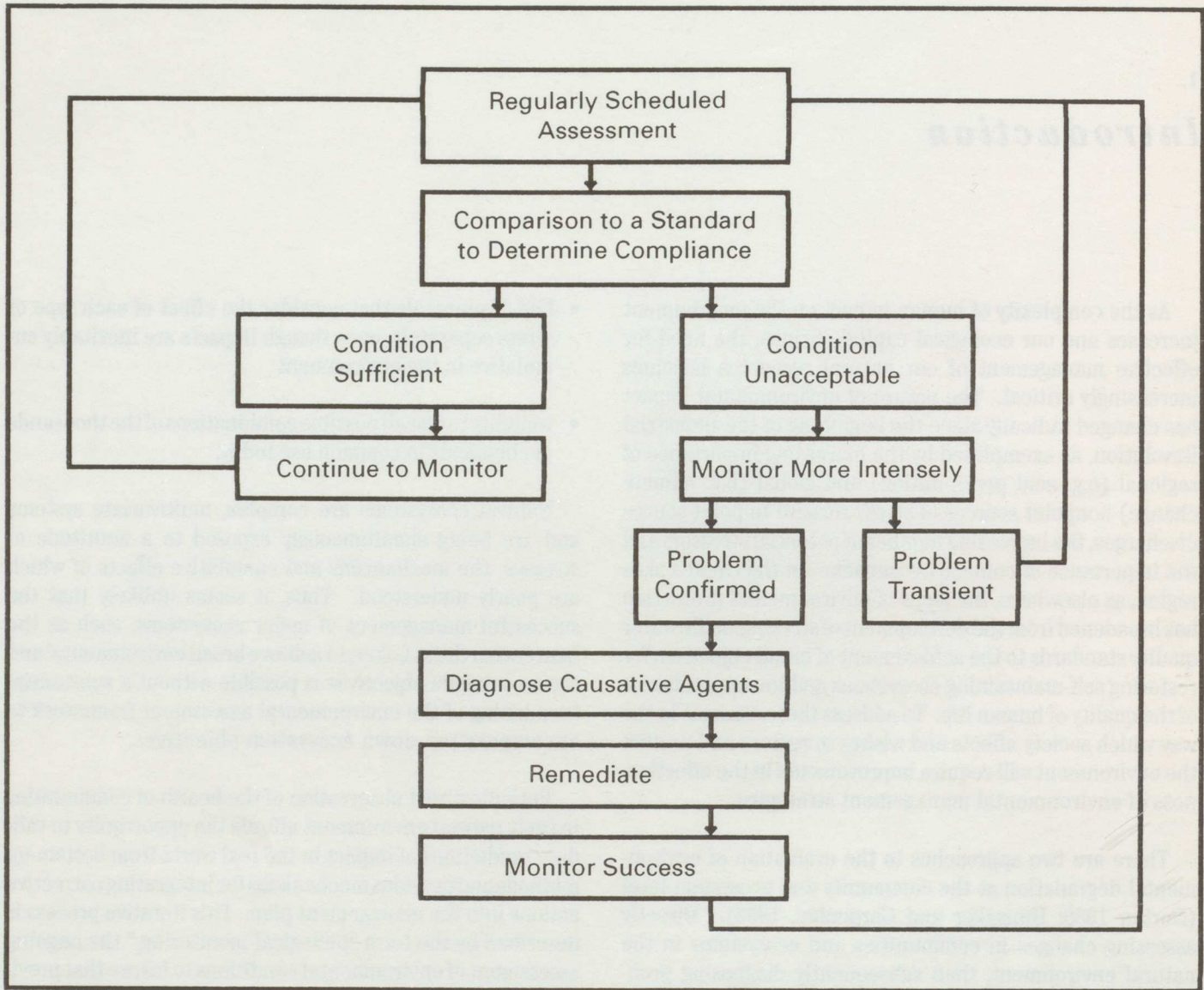


FIGURE 1. *Monitoring the Health of an Ecosystem Over Time.*

Above all, it is critical that the selection process be defensible. The importance of indicator selection cannot be overemphasized since any long-term monitoring program will be only as effective as the indicators chosen.

The focus of this report is on the development of an objective framework for selecting indicators of environmental health in the context of a long-term monitoring program for the Great Lakes region. This framework is based on the ecosystem approach, first formalized by the Great Lakes

Research Advisory Board (1978). This approach is conceptualized by the view of "man-within-the system" as opposed to "the ecosystem-external-to-man." Since its inception, this former view has evolved steadily towards a fundamental desire to promote compatibility between and sustainability of both ecological and human systems in the region (Great Lakes Water Quality Board 1989). The proposed framework supports this emerging goal by addressing the development of both biogeochemical and socioeconomic indicators of environmental health and the linkages between them.

2.

Relating Indicator Development to Management Goals for the Great Lakes Region

Implementation of an effective monitoring program for the Great Lakes, or any region, is contingent upon the development of explicit, generally-accepted ecosystem conditions to be achieved and maintained (i.e. ecosystem objectives). These attributes are derived from policy and management goals, developed as a result of input from political, social and scientific spheres (e.g. see Bertram and Reynoldson, 1991). Indicators are selected that are useful in judging the extent to which specific objectives have been achieved (e.g. that selected quality control parameters have remained within an acceptable range). Thus, it is clear that indicators cannot be identified until goals and objectives are specified.

Historically, policy and management goals relating to environmental protection in the Great Lakes region and elsewhere have centered on reducing the level of pollution entering natural receiving systems. This "end-of-the-pipe" focus is exemplified by the Great Lakes Water Quality Agreement of 1972, which established chemical-specific objectives for reducing loadings and in-lake concentrations of many known or suspected toxic substances and phosphorus, the primary eutrophying agent in the Great Lakes. Although this approach was successful at reducing loadings, many problems are inherent in the use of chemical and physical aspects of water quality as the sole yardstick of environmental health, perhaps the greatest of which is the inability to directly link changes in chemical emissions with changes in ecosystem health. Because the goal of environmental management is to protect natural ecosystems and human health, it is essential that ecosystem health be defined in these terms as well.

In view of the limitations of the sole reliance on chemical-specific objectives, revisions of the Great Lakes Water Quality Agreement have increasingly emphasized a broader "ecosystem approach" to managing the Great Lakes (e.g. Great Lakes Water Quality Agreement 1978), one which recognizes the interrelatedness of biotic and abiotic ecosystem components, including humans, and the relationship between the lakes and their surrounding watershed. This approach mandates the development of ecosystem as well as chemical-specific objectives, the former being clearly stated ecosystem conditions, primarily biological, that are to be attained and maintained under the revised Agreement.

The principal goal of management derived from the ecosystem approach has been to restore and maintain "the chemical, physical, and biological integrity of the lakes and their surrounding basins so that beneficial uses are not impaired." In keeping with this goal, objectives previously developed for Lakes Superior (Ryder and Edwards, 1985) and Erie (Edwards and Ryder, 1990) have focused on maintaining a balanced, stable oligotrophic and mesotrophic ecosystem, respectively. The development of indicators related to this objective has centered on the identification of surrogate organisms, species which integrate critical physical, chemical and biological properties of the ecosystem and, thus, can be used to judge the relative health of the ecosystem. Key indicator species chosen for monitoring, the lake trout (Lake Superior) and the walleye (Lake Erie), were determined to be useful not only for gauging ecosystem health, due to their role as top predators in these ecosystems, but for judging potential impacts on human use, a factor linked to their commercial importance.

Management goals in the Great Lakes are currently undergoing further evolution as the "ecosystem approach" to management merges with the concept of sustainable development (e.g. Great Lakes Water Quality Board 1989; Bertram and Reynoldson, 1991). Broadly defined, sustainable development encompasses ecological, economic and social issues, all of which are interdependent but not necessarily compatible (Munn, in review). The potential for conflicting goals associated with each of these aspects of sustainability demands that, from a management perspective, the goals be considered together.

In view of the broad context of sustainable development, previous goals and objectives and, thus, the corresponding indicators, developed for the Great Lakes, appear to take too narrow a view of the system in at least two respects. First, little explicit recognition has been given in any of these programs to the broader social and economic issues of the region, beyond those related to a few extremely important, but limited, activities (e.g. commercial fishing and human consumption of tainted fish). Secondly, there is a need to determine how humans view natural ecosystems beyond economic and recreational considerations. In particular, the development of an environmental ethic that promotes individual accountability and a realization of our common future

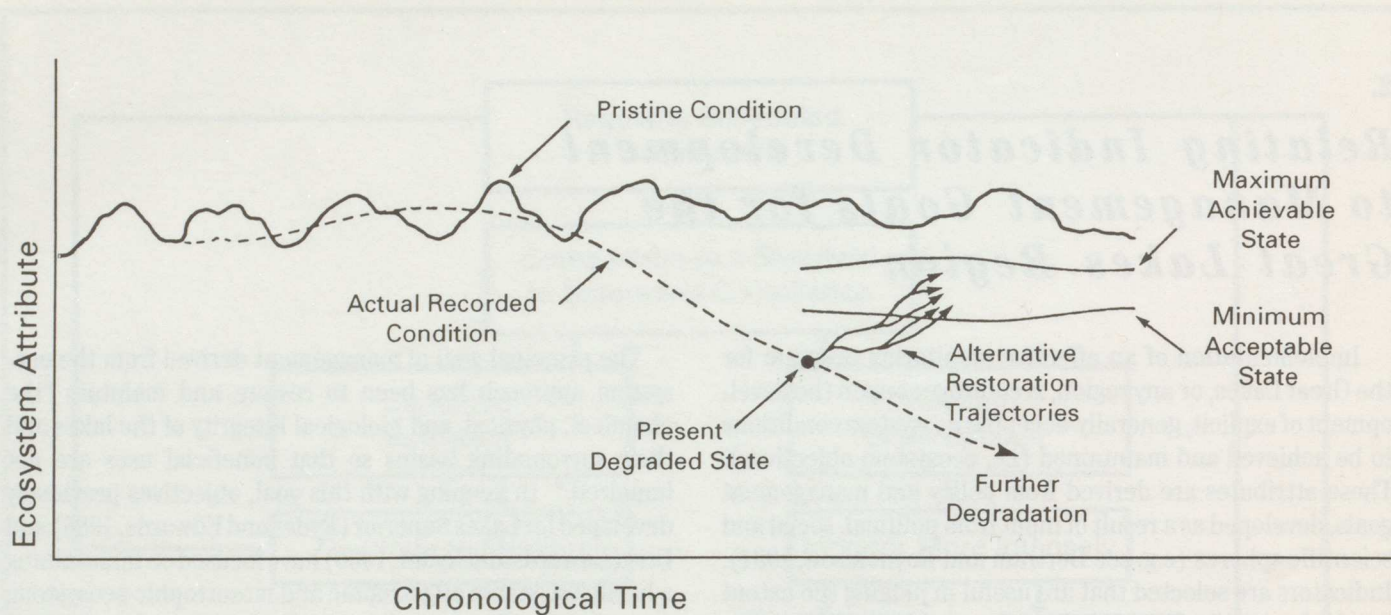


FIGURE 2. *Conceptual model of the trajectory of an ecosystem condition during recent history*

Several management strategies are currently in progress to restore the Great Lakes to some desirable set of conditions. While these efforts must meet some minimum acceptable set of criteria (i.e. chemical-specific and ecosystem objectives), progress towards a predisturbance condition is limited to some maximum achievable state by continued human impact.

will be critical to the achievement of sustainability. It should be noted that this second issue has begun to be addressed with the adoption of ecosystem goals and objectives for Lake Ontario (Ecosystem Objectives Working Group 1990). However, it is clear that a critical step towards addressing the emerging goal of sustainable development will be the adoption of a broad view of the concept of "man-within-the-system." Development of an indicator framework for supporting objectives proposed from this view is undoubtedly a critical emerging issue for Great Lakes managers.

There are three important concepts with regard to the management of the Great Lakes region in terms of sustainable development (e.g. Kerr 1990; Munn 1990):

- **Self-maintenance or self-sustainability of the ecological systems**, which means that the Great Lakes ecosystems have sufficient integrity that natural processes keep the quality conditions within an acceptable range through time, as expressed diagrammatically in Figure 2. It should be noted that self-maintenance is likely, even though the ecological condition of the ecosystems is not ideal, i.e. identical to that prior to extensive human settlement. Indeed, in some respects the lakes seemed to have retained certain properties associated with self-maintenance, even in their worst state (Allen 1990). This condition highlights the need to define and use the term "self-sustainable ecosystem" cautiously when stating ecosystem objectives.

- **Sustained use of the ecosystem for economic or other societal purposes**, which entails utilizing natural ecosystems for societal needs without degrading the resource. Sustained use is not possible if ecological capital is destroyed. Ecological capital might be the breeding stock of a commercially and recreationally valuable species, such as the lake trout. Another form of ecological capital is the pool of genetic information that has evolved over many thousands of years to make the structure and function of the Great Lakes ecosystems what is today or what it was in the relatively recent past. Reduction in the genetic pool through a loss of species richness may reduce the ability of the system to respond and adjust to future environmental changes, thus impairing the self-maintenance capabilities and efficient use of the resource. While it should be noted that sustained use may not necessarily require self-maintenance, the latter property is highly desirable since the subsidies necessary for sustained use may, be otherwise exceedingly expensive. For example, if the natural breeding stock or habitat is lost, hatcheries may be necessary to replenish stocks of juvenile fish and even sometimes adults, to maintain normal age recruitment and balance.

- **Sustained development to insure human welfare**, which includes not only medical issues relating to human health, but to broader issues concerning the potential for human development, including the perceived quality of life. This latter group of issues is probably one of the least considered from a management perspective.

While it is not the purpose of this report to propose future goals for the Great Lakes region, the considerations given above suggest the general goals that must be envisioned in order to move towards a sustainable future for the region. Indeed, some of the principal goals implied above have already been embraced (e.g. self-sustaining ecosystems). In this report, a principal message is that indicator development must proceed on a much broader scale than previously considered. Equally important is that policy and management goals are not static; even if the concept of sustainable development is adopted, the basic vision of sustainability for the Great Lakes region, as discussed above, will undoubtedly continue to change and be refined as society's view of environmental management and protection evolve. In this respect, caution must be exercised to ensure that changes in management goals will require that monitoring programs and indicator development be subject to ongoing review to assess the ability of these efforts to support stated objectives. Thus, a requirement of any framework for indicator development is that it be flexible enough to accommodate such changes in policy or management goals. This situation includes a recognition of not only impact assessment and ecosystem rehabilitation, but of impact **anticipation** and **prevention** as well.

The rest of this report will focus on a framework for selecting indicators that can be used to judge the attainment and maintenance of ecosystem conditions in the Great Lakes region compatible with the concept of sustainable development and the "ecosystem approach," as stated in the revised Great Lakes Water Quality Agreement of 1987.

While it is not the purpose of this report to provide a detailed analysis of the Great Lakes region, the committee does believe that the general public should be made aware of the need to have a national strategy for the Great Lakes. The primary purpose of this report is to provide a national strategy for the Great Lakes region. The report is organized into four main sections: a description of the Great Lakes region, a description of the current situation, a description of the proposed strategy, and a description of the implementation plan. The report is intended to provide a comprehensive overview of the Great Lakes region and to provide a clear and concise description of the current situation. The report is intended to provide a clear and concise description of the proposed strategy and to provide a clear and concise description of the implementation plan. The report is intended to provide a clear and concise description of the current situation, the proposed strategy, and the implementation plan.

The first of the report will focus on a description of the Great Lakes region. The report will provide a detailed description of the Great Lakes region, including a description of the Great Lakes basin, a description of the Great Lakes basin's geology, a description of the Great Lakes basin's climate, a description of the Great Lakes basin's vegetation, a description of the Great Lakes basin's wildlife, a description of the Great Lakes basin's human population, a description of the Great Lakes basin's economy, a description of the Great Lakes basin's culture, a description of the Great Lakes basin's history, a description of the Great Lakes basin's future, a description of the Great Lakes basin's current situation, a description of the Great Lakes basin's proposed strategy, and a description of the Great Lakes basin's implementation plan.

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3.

Framework for Developing a Monitoring Program

3.1 *General Rationale*

Once management goals have been specified, a framework must be developed for selecting indicators and utilizing the resulting information. *Basically, everything is an indicator of something but no one thing is an indicator of everything.* Economic and ecological considerations limit the number of indicators that can be measured to only a fraction of those available. Given such limitations, it is essential that indicators are selected in order to maximize unique, relevant information and minimize redundant information. The purpose of an indicator framework should be to organize the process of indicator selection and development, such that information is collected in a manner which is both cost-effective and most supportive of various management needs.

- We propose a framework for indicator selection that addresses three critical questions relating to ecosystem management:
- Are stated objectives being met?
- If stated objectives are not being met, what is the cause of this noncompliance?
- How can impending noncompliance be predicted before it is detected?

To answer these questions, a monitoring program must fulfill multiple purposes. The first and most obvious purpose is to provide an ongoing assessment of environmental conditions to determine if rehabilitation goals and objectives are being achieved, in terms of ongoing restoration efforts in the lakes, and if, once established, these conditions are being maintained (Figure 1; Hellawell 1978). Previous work on indicator development for the Great Lakes has focused heavily on this aspect of ecosystem management, ie. the identification of ecosystem parameters and processes that are useful for judging compliance with general goals and specific ecosystem objectives (e.g. Ryder and Edwards, 1985; Edwards and Ryder, 1990; Bertram and Reynoldson, 1991).

A second purpose of monitoring is to suggest corrective

actions in the event that objectives are not being met. Demonstrating the cause of environmental impact is a much more difficult task than merely observing that impact has occurred, and no single diagnostic method is suitable in all situations. However, certain general guidelines can be instructive. Gilbertson (1984) proposes a three-step diagnostic process:

- Identification of environmental impact
- Epidemiology, the process of determining the extent and nature of these effects and the formulation of causal hypotheses
- Etiology, which involves experimentation with the suspected stressor and other stressors known to exhibit similar effects, in order to reach conclusions regarding causation

These conclusions must be extensive enough to allow for rehabilitation strategies (e.g. Remedial Action Plans) to be formulated to correct the problem. Preferably, identification and diagnosis of a problem should occur early so that remedial actions can be taken before substantial damage has occurred.

It is highly unlikely that any single indicator can be found that fulfills all of the purposes stated above. In order to foster a comprehensive and organized approach to Great Lakes management, we propose development of an indicator program, based on three general types of indicators: **compliance indicators**, **diagnostic indicators** and **early warning indicators**. The rest of this section will be devoted to a description of the role which each of these performs in the proposed monitoring program. Desirable characteristics and specific criteria for the selection of each type of indicator will be covered in section 3.2.

Compliance indicators are those chosen to judge the attainment and maintenance of ecosystem objectives related to the restoration and maintenance of environmental quality in the Great Lakes region. While measures of compliance with chemical-specific objectives, namely the concentrations of the regulated substances themselves, are quite clear, the indicators most useful for judging the achievement of

broad ecosystem objectives (e.g. self-sustaining food webs) are not nearly as obvious. The most effective indicators of compliance with ecosystem 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), and there are several reasons to consider other biological parameters (e.g. community and ecosystem attributes) as well for indicators of compliance with ecosystem objectives. Compliance indicators will be the most obvious part of any monitoring effort and, thus, their significance should be readily communicable to the public and policymakers. Individual or population attributes of commercially and/or aesthetically important species (e.g. lake trout and bald eagle), for example, are useful as compliance indicators because effects on these species can easily be determined and communicated and because of other reasons. An example of a compliance indicator of economic condition might be the gross output of goods and services for the region.

In many cases those parameters most useful in judging compliance with a specified objective are not those best in determining why objectives are not being met. Causes of ecosystem deterioration are not always obvious or simple and, thus, may not be easily determined without an explicit protocol for addressing them. Diagnostic indicators, those parameters and processes that provide insight as to the cause of noncompliance, should be identified to facilitate this process. To a limited extent, the number of probable causative agents can be narrowed by correlating noncompliance with trends in other ecosystem or chemical-specific objectives. Information on changes in the quantity or quality of habitat or resources, or the water column concentration of a toxic chemical, for example, may be correlated with changes in levels of biological indicators (e.g. changes in lake trout population dynamics). Investigation of such correlative relationships has been proposed for use in other monitoring programs, including the Environmental Monitoring and Assessment Program (EMAP) currently under development at U.S. EPA (Hunsaker and Carpenter, 1990). Correlative relationships of this type are useful for the generation of hypotheses about potential causes, but alone do not provide strong evidence for cause-effect linkages, e.g. that which would warrant initiation of a costly Remedial Action Plan (RAP). The category of diagnostic indicators proposed here includes specific changes (e.g. enzyme changes induced by the bioaccumulation of a substance to toxic levels) that are capable of isolating specific stress effects on compliance indicators. It should be noted that not all diagnostic information need be gathered in situ; controlled laboratory and mesocosms testing to study etiology can be extremely useful in providing diagnostic information as well. Chemical fractionation of ambient water, followed by toxicity testing often provides useful diagnostic information (e.g. Mount and Anderson-Carnahan, 1988).

Together, the use of compliance and diagnostic indicators allows for reactive control when objectives fail to be met. Compliance indicators are used to determine that certain impacts have occurred or are continuing to hinder the achievement of ecosystem objectives. An appropriate suite of diagnostic indicators is then used to isolate the cause. This type of control relies on rehabilitation of the deteriorated state and, thus, tends to be quite costly and time-consuming. Furthermore, the success of such programs is often limited by a poor understanding of the functioning of ecosystems. Limitations associated with reactive management are exemplified by current problems associated with the development and implementation of Remedial Action Plans for areas of concern (AOCs) in the Great Lakes (Great Lakes Water Quality Board 1989). Such persistent problems have led to a call for predictive management programs within the region (Great Lakes Water Quality Board 1989).

The purpose of predictive management strategies is to identify impending problems before they exert substantial impact on the ecosystem. Compliance indicators will likely not be useful in this endeavor since these variables are chosen primarily to indicate the maintenance of some overt condition, which, once lost, generally requires substantial effort to restore (e.g. lake trout population levels). It is conceivable that, in many instances, noncompliance with ecosystem objectives can be predicted on the basis of laboratory tests or certain subtle changes (e.g. enzyme activity in individual fish) that respond rapidly to stressful conditions and anticipate changes of societal interest. Identification and surveillance of these early warning indicators of ecosystem change allow for management actions to be implemented before conditions have deteriorated to the point where compliance indicators are affected. When used in conjunction with diagnostic indicators, early warning indicators allow for the implementation of predictive management strategies. In view of the poor understanding of how ecosystems function and, therefore, how to rehabilitate them, the additional cost of monitoring early warning indicators in addition to compliance indicators to allow for predictive management may be a cost-effective alternative to a sole reliance on reactive management.

The overall indicator framework developed in this section is outlined in Figures 1 and 3. As developed above, this strategy encompasses efforts both to restore conditions that have been impaired by previous stressors and to prevent deterioration resulting from stressors that have yet to be identified and/or contained.

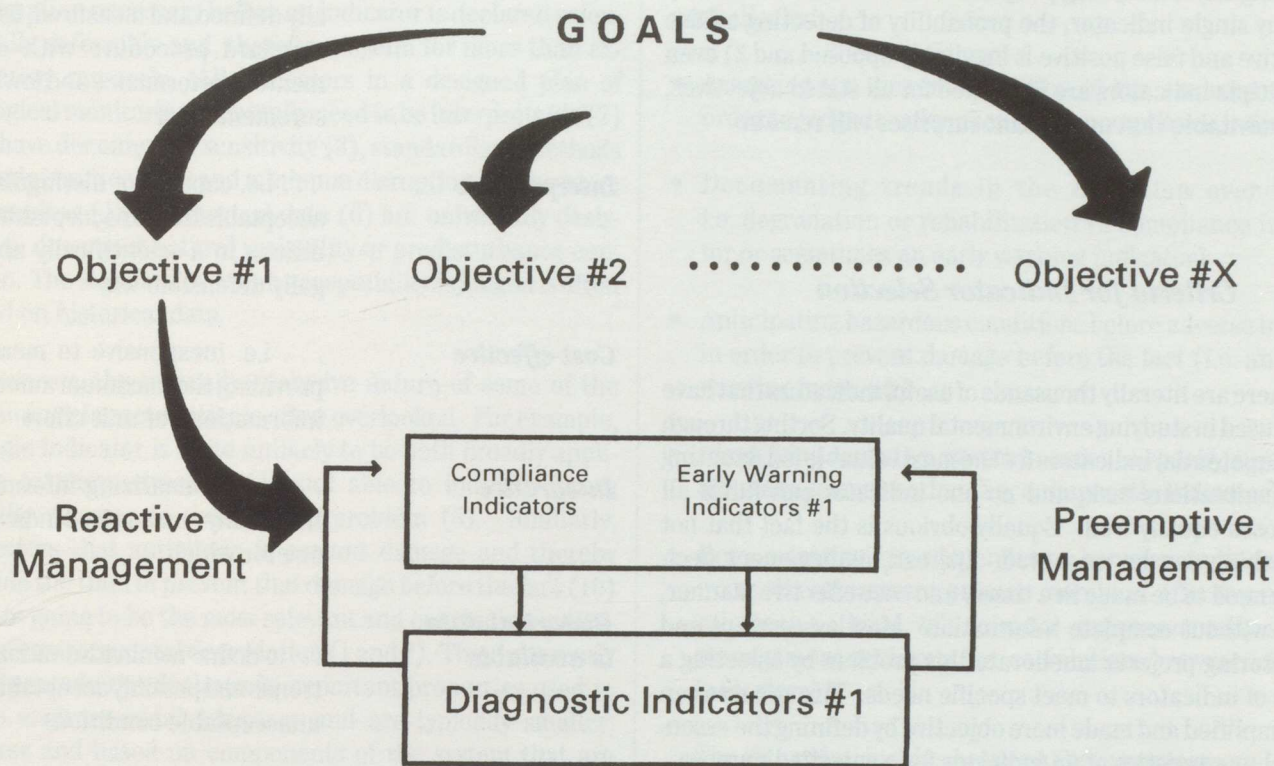


FIGURE 3. *General framework for indicator development*

Generally acceptable goals are used to develop a set of explicit ecosystem objectives. One or more compliance indicators are identified; these indicators are used directly to judge attainment and maintenance of some desired condition stated in an ecosystem objective. Early warning indicators are chosen to assist in maintaining the desired condition by detecting impending deterioration before substantial impact has occurred. Diagnostic indicators are essential for determining the management required for fulfillment of objectives.

No monitoring program, no matter how comprehensive or costly, can reasonably be designed to be infallible. One of us (Cairns) has been involved in 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 (National Research Council 1981), that attempted to work backwards from existing information to determine whether, with hindsight, it was possible to have predicted phenomena that had caused previous pollution problems of major consequence. The methyl mercury problem, for example, could have been predicted relatively easily if simple sediment-water microcosms had been included in the test protocol. Conversely, eggshell thinning in birds, resulting from DDT could not have been predicted by any available test models and procedures. Of course, other illustrations can be found to support conclusions on both sides. Exercises such as this one exemplify how certain "surprises" in ecosystem management may be predicted and, thus, be made amenable to pre-emptive action, while others may not be understood to the extent whereby preventive measures can reasonably be expected.

A primary focus of any monitoring framework must be to minimize the consequences of inevitable inaccuracies and uncertainties involved in ecosystem management. These uncertainties may take two forms: **1) false negative signals**, those that provide no warning of potential harm when, in fact, it is bound to occur and **2) false positive signals**, those that warn of potential harm when none, in fact, exists. Both are more likely to occur when too much reliance is placed on a single indicator or when the indicators selected leave large information gaps in the overall hazard evaluation or risk analysis process. Clearly, multiple lines of evidence are more likely to protect against unpleasant surprises by allowing for the validation of presumed positives and negatives. In this regard, **action levels**, predetermined responses gauged to address the level of urgency given a certain indicator response, can be developed to achieve an optimal cost-benefit ratio. The basis for such a program is simple: if only one of several indicators related to a specific objective suggests impairment, then little or no action may be required, while consistent responses across several indicators may require an appropriately gauged response. The feasibility of imple-

menting any monitoring program must realize two facts: 1) for any single indicator, the probability of detecting a false negative and false positive is inevitably opposed and 2) even if multiple indicators are identified for all stated objectives, it is inevitable that unpleasant surprises will remain.

3.2 *Criteria for Indicator Selection*

There are literally thousands of useful indicators that have been used in studying environmental quality. Sorting through all the potential indicators for the most valuable is a daunting and contentious task, and no one indicator can fulfill all purposes equally well. Equally obvious is the fact that not everything can be measured. Instead, management decisions need to be made in a timely and cost-effective manner, even without complete information. Most assessment and monitoring projects ameliorate this problem by selecting a suite of indicators to meet specific needs. This process can be simplified and made more objective by defining the essential characteristics of an indicator for a specified purpose.

Characteristics that are desirable in an indicator of environmental or water quality have been listed by various researchers (Suter 1989; Macek et al. 1978; Hammons 1981; Kerr 1990; Hunsaker and Carpenter, 1990). Ryder and Edwards (1985), Edwards and Ryder (1990) and Hellawell (1986) have developed similar lists, specifically focusing on desirable characteristics of a species chosen as an indicator of water quality. Despite the diversity of management problems that inspired these lists, there are several characteristics that are commonly mentioned. By paraphrasing, integrating and supplementing previous compilations, we have arrived at the following list. Ideal indicators would be:

- Biologically relevant** . . . i.e. important in maintaining a balanced community
- Socially relevant** . . . i.e. of obvious value to and observable by shareholders or predictive of an measure that is
- Sensitive** . . . to stressors without an all-or-none response or extreme natural variability
- Broadly applicable** . . . to many stressors and sites
- Diagnostic** . . . of the particular stressor causing the problem

- Measurable** . . . i.e. capable of being operationally defined and measured, using a standard procedure with documented performance and low measurement error
- Interpretable** . . . i.e. capable of distinguishing acceptable from unacceptable conditions in a scientifically and legally defensible way
- Cost-effective** . . . i.e. inexpensive to measure, providing the maximum amount of information per unit effort
- Integrative** . . . i.e. summarizing information from many unmeasured indicators, one for which
- Historical data is available** . . . to define nominative variability, trends and possibly acceptable and unacceptable conditions
- Anticipatory** . . . i.e. capable of providing an indication of degradation before serious harm has occurred, early warning
- Nondestructive** . . . of the ecosystem, one with potential for
- Continuity** . . . in measurement *over time*, of an
- Appropriate scale** . . . for the management problem being addressed. For the International Joint Commission, there are three relevant spatial scales: the Area of Concern, lakewide management and the basin ecosystem and many appropriate temporal scales
- Not redundant with other measured indicators** . . . i.e. providing unique information
- Timely** . . . i.e. providing information quickly enough to initiate effective management action before unacceptable damage has occurred.

Some of these characteristics summarize the background information necessary before an indicator is declared scientifically defensible and, therefore, useful for more than exploratory research. All indicators in a designed plan of biological monitoring eventually need to be interpretable (7) and have documented sensitivity (3), standardized methods (6), minimum cost (8) and minimum disruption to the system in sampling (12). Historical data (6) are universally desirable to document natural variability or predisturbance condition. The standard by which acceptability is judged is often based on historical data.

However, the mutually exclusive nature of some of the remaining characteristics is often overlooked. For example, a single indicator is quite unlikely to be both broadly applicable to many stressors (4) and able to indicate which specific stressor is causing the problem (5). Similarly, indicators that anticipate important damage and thereby provide the time to prevent that damage before the fact (10) are not going to be the most relevant and convincing indicators of environmental degradation (1 and 2). They inherently must precede the declines in important properties used to judge environmental adequacy and are typically smaller, quicker and based on components of the system that are valued less by the public and its representatives. Similarly, indicators that are good anticipators (11) are unlikely to be good integrators (9). A choice must be made between these characteristics. Differences in the ways indicators are used can guide the trade-offs necessary in selecting indicators

along these gradients. There are five purposes for which data are collected:

- Assessing the current condition of the environment in order to judge its adequacy (i.e. a compliance indicator)
- Documenting trends in the condition over time, i.e. degradation or rehabilitation (a compliance indicator or sometimes an early warning indicator)
- Anticipating hazardous conditions before adverse impact in order to prevent damage before the fact (i.e. an early warning indicator)
- Identifying causative agents in order to specify appropriate management action (i.e. a diagnostic indicator)
- Demonstrating interdependence between indicators to make the assessment process more cost-effective and to reinforce political will to make environmentally sound management decisions (i.e. correlations between various indicators).

The interrelationships between these purposes and their sequence in time are described generally in Figures 1 and 3. The characteristics of indicators that are particularly important for each specified purpose are ranked in Table 1. The characteristics are discussed below.

TABLE 1. Desirable characteristics of indicators for different purposes

Table entries are on a scale of importance from one to three, where one indicates lower importance and three indicates an essential attribute. Characteristics that are universally desirable and do not differ between purposes are marked with an asterisk.

Characteristic of Indicator	PURPOSE OF INDICATOR				
	Assessment	Trends	Early Warning	Diagnostic	Linkages
<i>Biologically relevant</i>	3	3	2	2	2
<i>Socially relevant</i>	3	3	2	2	2
<i>Sensitive</i>	*	*	*	*	*
<i>Broadly applicable</i>	2	2	2	1	1
<i>Diagnostic</i>	1	1	1	3	1
<i>Measurable</i>	*	*	*	*	*
<i>Interpretable</i>	3	3	2	1	1
<i>Cost-effective</i>	*	*	*	*	*
<i>Integrative</i>	2	2	1	1	2
<i>Historical data</i>	*	*	*	*	*
<i>Anticipatory</i>	1	1	3	1	2
<i>Nondestructive</i>	*	*	*	*	*
<i>Continuity</i>	2	3	1	1	1
<i>Appropriate scale</i>	*	*	*	*	*
<i>Lack of redundancy</i>	*	*	*	*	*
<i>Timeliness</i>	2	2	3	3	2

Indicators for the assessment of current ecosystem condition and judgment of adequacy (compliance objectives) need high biological and social relevance in order to be effective at documenting the health of the environment in terms that are understandable to the shareholders. Readily understandable information will be most likely to encourage appropriate management action. These indicators also need to be readily interpretable so that measurements requiring management action are predetermined and scientifically justifiable. This condition may require that an historical data base be available to provide information on original condition and natural variability or that measurements on healthy, reference ecosystems be available for comparison. Broad applicability to different stressors would permit standardization across Areas of Concern and increase the likelihood that an indicator would also reflect changes in environmental health due to new and unanticipated stressors.

Indicators for the assessment of trends over time are used to document recovery in response to Remedial Action Plans and to monitor ecosystem health over the long term. These indicators also need high biological and social relevance and broad applicability. In many cases, the same indicators are used for assessment and for trend monitoring. But, indicators for monitoring trends would ideally have additional characteristics. Lack of continuity, which is the ability to measure the same response over a long time period, compromises the ability to identify real changes in the ecosystem over time. For example, a lack of continuity due to changes in detection limits has complicated much of the interpretation of the levels of toxic substances in the Great Lakes. Reliance on a single species to monitor effects on lower trophic levels is also problematic in terms of continuity. If the species were replaced in the ecosystem by a functionally similar organism, the ecosystem could persist relatively unchanged, but trend monitoring would be compromised. In addition, historical information about natural variability over successional time frames may be important for indicators used in monitoring trends.

Anticipation of unacceptable conditions, before the fact, is a special case of trend monitoring which focuses on prevention of an adverse effect. When an environmental effect is particularly undesirable, we do not want to wait for damage, document it and remediate it. Instead, we wish to avoid the damage. When an indicator signal can precede significant damage, it is possible to preempt the damage through immediate intervention. The best indicators for early warning are usually quite different from those for assessment. While relevance is of paramount importance for assessment, and continuity is important in most trend monitoring, timeliness is of paramount importance for the anticipation of effects. The indicator must respond, be measured, be interpreted and initiate management action in sufficient time to head off significant damage. In demanding a quicker response to

stressors and lead time before unacceptable effects, early warning indicators will be smaller, quicker and of less immediate value to shareholders.

Diagnostic indicators are used to isolate probable causative agents after a problem has been identified and to prescribe appropriate management actions. Without a means of isolating the causative agent, management responses are unlikely to be efficient and cost-effective. Diagnostic indicators must single out causes rather than integrate them. These indicators follow the general assessment of ecosystem conditions and, in contrast to them, may be very site-specific and reductionist rather than broadly applicable.

Linkages between the biogeochemical and socioeconomic spheres will most likely be demonstrated through the correlation of appropriate indicators rather than through the use of unique indicators. The establishment of strong relationships between these two areas may make a particular indicator more valuable, as its biological and social relevance will increase, as will its cost-effectiveness. The effects of human activity on the ecosystem have been well documented over the years. However, the adverse effects of ecosystem degradation on human activity has been less documented and undervalued. By exploring relationships between biogeochemical indicators and socioeconomic indicators, the interconnectedness of humans and their ecosystem can be incorporated into the monitoring plan.

The choice of indicators from the thousands available can proceed hierarchically. First, the indicator must be intimately related to management goals. The indicator should be logically related to the management decision (Suter 1989) or closely related to another indicator that is. Because management goals embody current interpretations of social and biological relevance, the indicators resulting from close ties to management goals will be relevant. Second, the appropriate temporal and spatial scales of the endpoint will be dictated by the management goal. For example, if the goal is early warning of human health effects, a quick response of a cellular receptor sampled on the same scale as the drinking water supply might be called for. If, instead, the goal is assessment of the state of free ranging sport fishery stock, a response integrating the entire life-cycle of the receptor organism sampled across its habitat would be called for. Third, relatively few indicators are sufficiently developed to present no major methodological problems or challenges to their scientific or legal defensibility at this time. But some indicators are more defensible than others or more promising for future development.

Evaluating Available Monitors

4.1. *Introduction*

The purpose of this section is to establish priorities for groups of candidate indicators, based on available information and using the general objectives and framework developed in Sections 2 and 3, respectively. As stated in the introduction of this report, indicator identification is based on the goals and objectives set for a particular ecosystem or region. This statement motivates a very broad interpretation of the term "indicator." According to this scheme, for example, concentrations of toxic substances may be termed an indicator (compliance) of the achievement of chemical-specific objectives outlined in the Great Lakes Water Quality Agreement (GLWQA). Indicators of ecosystem and human health are generally not so obvious and, ultimately, their selection will be based on the ecosystem objectives to be met. Recommendations made in this section are tentative, given that explicit ecosystem objectives are still being formulated, but assume that future objectives will relate to broad environmental and social goals of sustainability.

Physicochemical, biological and socioeconomic (including human health) indicators are considered separately in this section. This categorization reflects traditional approaches to environmental monitoring. Linkages between different categories of indicators have not been well established, but are discussed within each of these sections, but particularly in the final section on socioeconomic indicators, with regard to the relationship between environmental and human effects. Recommendations for further research in this area are made in the final Section 5.

4.2 *Physicochemical Indicators*

Changes in the concentration of both natural and xenobiotic chemicals have had a profound and, in many cases, well-documented effect on ecological and human processes in the Great Lakes. Ecosystem subsidies resulting from the increased availability of nutrients (e.g. phosphorus) affect ecosystem operation by stimulating primary productivity, altering the taxonomic composition and food quality of

primary consumers and increasing community respiration with concomitant reductions in oxygen levels. Loadings of toxic contaminants (e.g. PCB, PAH and heavy metals) impair population-level processes (i.e. growth and reproduction), thereby altering community structure and ecosystem function. Chemical stressors elicit effects on humans, both directly (e.g. drinking or swimming in contaminated water or breathing contaminated air) and indirectly through effects on other biota (e.g. eating contaminated fish or causing aesthetic problems, such as noxious algal blooms).

Recognition of the effects of chemical conditions on the "health" of the Great Lakes is illustrated by the strict and extensive guidelines concerning the emission and bioavailability of known or suspected toxics in the lakes. Monitoring of several chemicals is already required to assess compliance with chemical-specific objectives stated in the Great Lakes Water Quality Agreement. Adherence to specific chemical criteria is essential in order to improve water quality and, thus, biological conditions in the ecosystem. However, there are several problems that limit the usefulness of this type of testing in judging compliance with ecosystem objectives (Wall and Hanmer, 1987):

- It is impossible to monitor concentrations of all chemicals, given cost and technological constraints
- All potentially toxic chemicals are not known
- Knowledge of chemical concentrations in water does not always provide an accurate picture of biological availability
- Chemicals may react synergistically and antagonistically with each other and with other environmental factors (e.g. hardness concentration of water)

The shift of the United States Environmental Protection Agency away from a sole reliance on chemical testing reflects these concerns (U.S. EPA 1985). In short, while a necessary part of any comprehensive monitoring program, routine chemical analyses are unreliable predictors of ecosystem health.

In general, measurements of water quality conditions will only serve as useful early warning indicators in instances where a specific chemical culprit is suspected. The use of

chemical parameters in this manner will be most useful on small spatial scales, such as point source discharges or, possibly, Areas of Concern, where specific chemicals have been targeted and monitoring is being conducted with relatively high frequency. By the time toxic chemicals reach detectable concentrations at the basinwide level, substantial biological impact has probably already occurred.

Measurements of water quality parameters that indicate changes in environmental conditions important to the biota (e.g. dissolved oxygen, pH, conductivity, nutrients) will be similarly limited in their usefulness as early warning indicators of environmental change. In most cases, by the time changes in these conditions are detected, substantial insult to the ecosystem has already occurred. For example, by the time declines in oxygen concentrations occur in the hypolimnion of a lake, loading of stimulatory substances has already occurred to the extent that several biological changes have already occurred (e.g. elevated primary productivity); action at this point may not suffice to avert further damage (e.g. fish kills). Local changes in water quality may be useful in forecasting trends on larger spatial scales (e.g. basinwide) if sampling is properly designed and implemented. Use of data from Areas of Concern would not be useful in this regard because changes in water quality due to site-specific remediation efforts would likely not be predictive of basinwide trends.

While not always reliable in predicting biological responses, chemical measurements are essential for diagnosing the cause of changes in biological parameters. Field surveys documenting biological effects must be accompanied by evidence that a suspected chemical stress is present in the affected location(s). Of course, such correlative information alone is inadequate for establishing a causal link; changes may be due to concomitant changes in the environment other than chemical pollution. The use of controlled testing, either laboratory or field, is required to establish such a cause-effect linkage.

As with chemical indicators, changes in physical attributes of the ecosystem (e.g. water level and temperature, turbidity, sedimentation) will generally be most useful for diagnostic purposes. Habitat assessments (e.g. Plafkin et al. 1989) are essential in evaluating causes for biological declines. Habitat parameters are selected that relate to overall use by aquatic life. For benthic habitats, suitable measures might include bottom substrate composition and stability, presence of suitable cover or refugia and degree of siltation. Assessments are compared to a "reference" or "best attainable" situation.

Because much of the change in chemistry and physical structure can be attributed to human activities, many systems have been developed to measure stress intensity by

quantifying and summing classes of human activity that produce loadings to the system (Leonard and Orth, 1986; IJC 1989; Hunsaker and Carpenter, 1990). Factors such as population density, miles of road, number of dwellings, point-sources of discharge, use of pesticides and land-use changes (e.g. wetland loss, clearcutting) are included in these calculations.

4.3 *Biological Indicators*

4.3.1 Introduction

Biological responses tend to integrate the independent and interactive effects of many stressors, a property that makes them more robust indicators of ecosystem condition than the concentrations and loadings of individual chemicals. Indeed, only biological material can be used for indicating the effects of chemical stressors in an ecosystem. Methodologies for the measurement of some chemicals discharged into surface waters are not well developed, and toxicological information is unavailable for many more. In the United States, these realities are reflected in a movement away from a sole reliance on chemical-specific environmental monitoring to an approach that includes biological-based evaluations of hazard and environmental condition (U.S. EPA 1985; Wall and Hanmer, 1987; Hunsaker and Carpenter, 1990).

Ecosystem objectives developed for the Great Lakes will undoubtedly require identification of biological parameters that can serve as key compliance indicators. Management efforts will be greatly aided by the identification of other biological parameters that can function as early warning and diagnostic indicators to complement compliance indicators for each of these objectives. Because some of the characteristics deemed desirable for different types of indicators are incompatible (see Table 1), it is unlikely that any single measure will be ideal for all purposes.

Although the focus of this discussion on biological indicators is on field surveillance, the role of experimental bioassay techniques in future monitoring programs is first determined. Laboratory and field experimentation provides strong evidence for cause-effect linkages under conditions that are typically low in environmental realism. Field observation and monitoring allow for direct assessment of exposure and effects, although it is often difficult to establish cause-effect linkages.

Analysis of in-situ indicators of an ecosystem's condition is considered in two general classes of biological organization: 1) measurements performed on individuals or popula-

tions of specific target species and 2) measurements performed to assess community/ecosystem structure and function. Measurements of population, community and ecosystem levels tend to be more appropriate compliance indicators for judging the achievement of ecosystem objectives, which will likely focus on issues such as the sustainability of target populations and the larger lake community. Conversely, measurements performed on individuals (e.g. enzyme analyses) will tend to be better diagnostic and early warning indicators. Stressors tend to affect biota at lower levels of biological organization (e.g. effects of persistent contaminants on biochemical and physiological processes of individuals) and subsequently affect parameters at higher levels (e.g. commercial fish yield), that are valued by society. Thus, a comprehensive monitoring program, for example that proposed in Figures 1 and 3 in Sections 1 and 2, will undoubtedly require the use of measures at several levels of biological organization.

Finally, the utility of different biological measures must also involve a comparison of different taxonomic groups. In this report, the major taxonomic groups considered are operationally defined, based on the types of databases available, for example terms such as the zoobenthos refer to communities that encompass several taxonomically-distinct groups (e.g. annelids and arthropods) that, for the purposes of most applied research endeavors, are studied together.

4.3.2 The Role of the Bioassay

Several syntheses are already available that discuss the role of controlled laboratory and field experimentation in environmental assessment (Cairns 1985, 1986b; Rand and Petrocelli, 1985; Cairns and Niederlehner, 1987; La Point et al. 1989; La Point and Perry, 1989; Cairns and Mount, 1990; Côté and Wells, 1991) and, therefore, details of their use will not be repeated here. In short, a vast number of alternative experimental designs are available for conducting controlled tests, and the relative utility of specific designs varies greatly with the circumstances under which environmental assessments are made and the types of stressors being studied.

Certainly, a sole reliance on bioassays for determining environmental management policies would be unwise. While bioassays vary greatly in their environmental realism (e.g. single species laboratory tests vs field mesocosms containing whole communities), any controlled study will de facto involve some departure from completely natural conditions. This reality alone dictates that experimental predictions be confirmed in the real world, although this proposition is often not simple. Criticisms of this type have targeted the use of simple laboratory bioassays as the foundation of the "bottom-up" view of environmental toxicology (e.g. National Re-

search Council 1981; Kimball and Levins, 1985). In the context of the present discussion, it is argued that laboratory and field bioassays can often be integral parts of diagnostic processes used to determine specific causes of observed or impending environmental impacts. Just as the control measures incorporated into experimental bioassay designs reduce the ability of these tests to accurately predict potential environmental effects (although conservative estimates of actual effects based on such predictions certainly aid in avoiding impact), the ability to directly test hypotheses regarding the causes and mechanisms of environmental impact enhances the utility of this element of hazard assessment and remediation.

Laboratory and field bioassays are equally useful as diagnostic tools for the restoration of desirable ecosystem conditions and as predictive tools for preventing environmental impact. Controlled experimentation is often an important element of efforts to rehabilitate sites affected by anthropogenic stressors (Jordan et al. 1987; Cairns 1988). Similarly, failure to achieve restoration goals and objectives will require that a diagnostic procedure be implemented to determine exactly why previous management strategies were ineffective and how future efforts should be refined. Finally, laboratory and field bioassays will continue to be the basis for predicting the potential hazard of recent or impending threats (e.g. unlicensed chemicals) to the environment.

Bioassays vary greatly in their complexity, both in terms of the level of biological organization examined and the inclusion of realistic environmental conditions into their design. The single species bioassay (Rand and Petrocelli, 1985; Côté and Wells, 1991) remains the backbone of laboratory hazard evaluation procedures, despite certain inherent limitations (e.g. National Research Council 1981; Kimball and Levins, 1985; Cairns and Niederlehner, 1987). The development of test procedures at the community and ecosystem levels, using controlled laboratory and field test systems (i.e. microcosms and mesocosms) offers an alternative or, in many cases, a complementary approach to single-species bioassays (e.g. compendiums such as Giesy 1980; Cairns 1985, 1986b). Traditional "bottom-up" approaches to hazard assessment incorporate bioassays at various levels of complexity into specific tiers of testing. In contrast, "top-down" approaches to diagnosis are largely *ad hoc*. However, while it remains difficult to describe generic protocols for such monitoring programs, the usefulness of such procedures is increasingly recognized.

4.3.3 Measurements on Individuals and Populations

The basis for this approach is the selection of species that provides interpretable indications of changing environmen-

tal conditions. Measurements performed on these species may indicate exposure to a stressor (e.g. bioaccumulation) or effects resulting from exposure (e.g. increased incidence of carcinogenesis). Candidate indicators of environmental stress considered within this broad category include:

- **Biochemical effects** at the cellular and subcellular level (e.g. enzyme induction)
- **Body burdens** of chemicals in various tissues of individuals, used as an indicator of exposure
- **Growth rate** of individuals
- **Carcinogenesis**
- **Teratogenesis** and congenital defects
- Susceptibility to **disease**
- **Behavioral** effects
- **Morphological changes** in algal cells, etc.
- **Feminization**
- **Abundance and biomass** of individuals in the population
- **Production** or yield
- **Natality and mortality**
- **Population age structure**
- **Population size structure**
- **Number of breeding pairs**
- **Geographical range of population**

These parameters have been measured with different species and, obviously, not all parameters are applicable to every taxonomic group. As discussed below, individual species or measures may be relatively sensitive to certain stressors and extremely insensitive to others. A suite of indicator species may be necessary to provide a comprehensive assessment of changes in an ecosystem's condition related to a multitude of important stressors. Species that are complementary in terms of their sensitivity to various stressors should be identified for this purpose.

Changes in the biochemistry of individual organisms are the basis for many effects at higher levels of biological organization. Alterations in molecular structure (e.g. ge-

netic effects), immunological responses and enzymatic activity can subsequently exert significant effects on the growth and survival of individual organisms and, indirectly, on the dynamics of populations and communities. These "biomarkers" are increasingly being recognized as powerful diagnostic and early warning indicators in environmental monitoring, and research in this area is proceeding at a rapid rate (DiGiulio 1989; McCarthy and Shugart, 1990). Stress-specific changes at this level are useful as diagnostic indicators. Generic responses to stress at the biochemical level, that can be related to effects at the individual and population levels, can serve as useful early warning indicators of stress (i.e. the clinical indicators of Giesy et al. 1988). As stated previously, the rationale for expanding existing monitoring programs to include such early warning signals arises from the reality that it is easier (and less costly) to prevent impact than to attempt to restore after impact.

Sublethal physiological and behavioral changes in individual organisms related to stress are the basis for many laboratory-based bioassay and monitoring protocols. Various attributes serve as response indicators of acute and chronic stress. Chemical burdens in tissues are frequently used as indicators of exposure. Changes in vital signs (e.g. respiratory rate) are extremely sensitive early warning indicators of stress for laboratory monitoring (e.g. on-line monitoring of effluent quality, Cairns and Gruber, 1980), but are not easily measured in the field. Although not as pre-emptive, outward signs of individual condition (e.g. disease or tumors) can be used for field assessments of the condition of larger organisms such as fish (Karr et al. 1986). Monitoring programs utilizing organismal responses have traditionally used selected fish and mammal species. More recently, tests using lower organisms (e.g. the Microtox assays of Dutka et al. 1983, that measure bacterial cell fluorescence) have also been shown to be useful hazard assessment and management tools. Measures of attributes of individual organisms may serve as compliance indicators (e.g. body burdens of toxic contaminants) or as diagnostic tools (e.g. teratogenic effects). Because organismal responses usually emanate from biochemical changes, the latter are generally more sensitive early warning indicators.

Population-level parameters are commonly used as assessment endpoints in the laboratory and field settings to measure the effects of stress. Measures of abundance and production may be useful compliance indicators both for commercially valuable species (e.g. maintenance of a certain annual yield of lake trout) and nuisance species (e.g. maintenance of sea lamprey density below a certain level). Other measures are somewhat more diagnostic (e.g. estimates of reproduction and mortality or age structure). For species that have been extirpated from portions of the region as a result of deteriorated environmental conditions (e.g. bald eagle), the expansion of geographical distribution or changes

in the number of breeding pairs may be useful as compliance indicators for gauging the success of management efforts related to certain restoration objectives. Although the measurement of most indicators requires that formal monitoring programs be established, certain population-level indicators can be measured with public participation (e.g. Christmas bird counts to estimate changes in population size and geographical range of rare species).

Terminal predators are the most widely-supported candidate indicator species for assessing environmental conditions, largely because of their susceptibility to persistent toxic contaminants, which are magnified as they move up through the biological food web and because of the commercial and aesthetic value placed on many such species. The three taxonomic groups that include top predators (fish, birds and mammals) have different attributes that recommend either for or against their use as indicators. For example, while wild mammals (e.g. mink) may be superior for predicting potential health consequences in humans, they are difficult to monitor because of their elusive habits. Predatory fish such as the lake trout are economically important but may be rather poor predictors of effects on humans.

Using criteria similar to those proposed in this report, Ryder and Edwards (1985) recommended the lake trout as the optimal indicator species for measuring environmental conditions in oligotrophic (low productivity) ecosystems. Similarly, the walleye was chosen as the primary indicator for gauging the recovery of habitats that were historically mesotrophic (moderate productivity) (Edwards and Ryder, 1990). In addition to their position as top predators in the aquatic food chain, the suitability of these species is enhanced by a thorough understanding of their autecology and their ability to act as "integrator organisms," one which reflects both direct and indirect effects of various environmental stressors (Ryder and Edwards, 1985).

In recommending a single species for monitoring ecosystem conditions, it was recognized that no "ideal" indicator organism exists. Companion indicators were, therefore, chosen for both oligotrophic and mesotrophic conditions. The benthic amphipod, *Pontoporeia hoyi*, was considered to be a suitable complementary oligotrophic indicator species to the lake trout; both its location within the ecosystem and its relative sensitivity to different types of stress are somewhat different from that of the lake trout (Ryder and Edwards, 1985). A second member of the zoobenthos, the insect *Hexagenia limbata*, was chosen as a companion indicator to the walleye in mesotrophic habitats (Edwards and Ryder, 1990). Identification of indicator species for Lake Ontario has also focused on *Hexagenia* (Reynoldson et al. 1989).

The above choices for indicator species appear generally sound. The selection of complementary indicators (i.e. those

that integrate somewhat different aspects of environmental stress) is particularly useful. Because individual species may differ substantially in their relative sensitivity to different types of stress (e.g. Patrick et al. 1968; Mayer and Ellersieck, 1986), additional species may be required to produce a monitoring program that is sufficiently robust to address broad ecosystem objectives, such as those currently under development for Lake Ontario (Bertram and Reynoldson, 1991). In view of the environmental and toxicological complexity of conditions in the Great Lakes region, it is doubtful whether one or two species will be sufficiently sensitive to all major impacts affecting these ecosystems. For example, Friend and Rapport (1990) note that, while lake trout may be rather sensitive to the effects of toxic contaminants and eutrophication, it may not be particularly sensitive to stresses occurring primarily in nearshore areas (e.g. wetland fragmentation and loss) or in tributaries of the lakes (e.g. acidification). Similarly, while *Hexagenia* appears to be quite responsive to changes in benthic oxygen concentration caused by eutrophication (Reynoldson et al. 1989), the sensitivity of this species to other forms of environmental change is unclear.

As with predatory fish such as the lake trout, a reasonably comprehensive database exists for a second set of terminal predators, fish-eating birds. The devastating effects of pollution, particularly persistent organic contaminants (e.g. organochlorine pesticides), on a number of avian species have been well documented (reviewed in Gilbertson 1988; Peakall 1988). Many useful measures of organismal and population stress have been proposed for species such as the herring gull. Population measures, such as geographical distribution and the number of active breeding pairs for more sensitive species (e.g. the bald eagle and the osprey), may serve as indicators of compliance with ecosystem objectives related to a "healthy" ecosystem. The potential for public participation in certain monitoring activities (e.g. bird censuses) exists for these measures, which generally rank well in terms of public appeal. A number of biochemical and physiological indicators of stress have been developed that serve diagnostic and early warning functions (e.g. Gilbertson 1988; Peakall 1988). The extent to which these species integrate the impacts of different forms of anthropogenic stress appears to be less than that purported for species such as lake trout. Indeed, while habitat fragmentation may have contributed to reductions in the abundance of certain bird species (e.g. Forster's tern), the major stressor implicated in the decline of populations of piscivorous birds are organochlorine pesticides, which persist in several ecosystem compartments. Although this inordinant effect of one type of stressor may reduce the usefulness of piscivorous birds as general indicators of stress, it has the benefit of allowing for monitoring of the effects of one of most troublesome groups of persistent contaminants in the lakes.

The herring gull has been the most intensively studied species with respect to the impact of pesticides on the Great Lakes. The usefulness of this species has been questioned for several reasons (Ryder and Edwards, 1985):

- **Environmental tolerances too broad**
- **Opportunistic feeding habits**
- **Seasonal migration between the upper and lower lakes**
- **Lack of standardization of commonly used measures (e.g. reproductive success)**

These criticisms notwithstanding, the herring gull appears to rank quite high as a candidate indicator species. The herring gull is a widely distributed terminal predator, and populations exhibit a year-round presence in the lakes region (Gilman et al. 1979). The major reason for not fully considering the herring gull as a principal indicator appears to be related to the relatively broad tolerance of this species for various environmental factors. However, while it is true that population-level parameters for this species may be somewhat insensitive to stress, a number of sensitive biomarkers and morphological effects have been developed and widely used, as described above.

Unquestionably, there are advantages to using more stenocious species, such as the bald eagle or the osprey that engender more public concern than do gulls, and, certainly, monitoring the recovery of these two populations (e.g. number of breeding pairs, changes in geographical range) may be useful in assessing compliance with broad ecosystem objectives for maintaining indigenous populations. However, the extensive database available for the herring gull and the host of available measures of environmental effects make this species immediately available as a monitoring tool in the lakes region.

Other candidate indicator species may exhibit selective sensitivities to other types of stresses in the Great Lakes. Fish-eating mammals such as mink, for example, appear to be particularly sensitive to polychlorinated biphenyls (PCBs), another class of toxic contaminants (e.g. Aulerich and Ringer, 1977; Harris 1988). The phylogenetic similarity between these species and humans makes them potentially useful indicators for assessing human health as well as environmental effects. However, these species are typically rather difficult to monitor and, consequently, autecological and distributional information is somewhat sketchy as well. Sensitive indicators of PCB-induced stress (e.g. enzyme induction or other biochemical changes) are not well developed for these species (Fitchko 1986). Thus, these species would become less useful as a monitor of PCB contamination since concentrations of these substances in the environment de-

crease to levels where gross effects no longer occur, but biochemical effects may persist and concern should still exist.

Populations of primary producers are an obvious choice for monitoring changes in the extent of eutrophication due to phosphorus loading into the lakes since they are the biological interface between changes in phosphorus availability and ecosystem impacts. Algae are the dominant primary producers in the Great Lakes and, thus, occupy an important position in aquatic food webs. Although undoubtedly affected by toxic materials in the lakes, algal species have been influenced most strongly by changes in phosphorus loading in the lakes following human settlement (Sicko-Goad and Stoermer, 1988). This phenomenon makes them a particularly good set of indicators for tracking changes in phosphorus availability and nutrient limitation in the lakes.

The macroscopic chlorophyte, *Cladophora*, has been proposed as an indicator of phosphorus loading (Auer et al. 1982). The abundance of *Cladophora* in the Great Lakes has increased dramatically in response to increases in available phosphorus in nearshore areas, and prolific growths of this taxa can significantly impair beneficial uses in these areas (e.g. use of public beaches) (Auer et al. 1982). Species such as *Cladophora glomerata* generally grow attached to the substrate, thus making them good integrators of local environmental conditions over time. Mean availability of phosphorus over long periods of time can be estimated from internal phosphorus concentrations (Auer et al. 1982). Simple measures, such as areal *Cladophora* biomass, are useful for quantifying point source phosphorus loadings, but must be performed frequently since substantial sloughing of algal growth may occur during storm events. Because of certain methodological and species-level taxonomic difficulties, alternative species, such as epiphytic algae growing on *Cladophora*, may be more useful indicators if developed further (E. F. Stoermer, University of Michigan, Pers. comm.)

The United States Environmental Protection Agency has opted for a nontaxonomic measure of nutrient availability and eutrophication in developing its Environmental Monitoring and Assessment Program (EMAP) (Hunsaker and Carpenter, 1990). This trophic state index (TSI) is based on measurements of chlorophyll-*a*, water clarity (e.g. Secchi disk transparency) and total nitrogen and phosphorus in the water column. The advantage of this index as an indicator of eutrophication is that it is easy to measure. Measurements utilizing attached algal species, such as *Cladophora*, allow for more localized assessments of phosphorus availability, whereas measurements performed on the plankton are more appropriate for lakewide assessments. To some extent, therefore, these measurements may provide complementary information. Measurements such as chlorophyll-*a* may be affected not only by bottom-up trophic effects caused by changes in



Herring gulls are used as a diagnostic indicator of contamination in the Great Lakes.

phosphorus availability, but by top-down (i.e. herbivory and predation intensity) effects as well. Changes in planktivorous fish abundance, for example, may affect the chlorophyll-*a* standing crop in the water column of lakes by altering the abundance and composition of herbivorous zooplankton assemblages (Carpenter et al. 1985). The introduction of the benthic exotic species, the zebra mussel, has been implicated in increased water clarity in Lake Erie (Roberts 1990), an influence which is unrelated to changes in phosphorus availability. In contrast, *Cladophora* is not used as a major food source by any organism. The relative efficacy of taxonomic and nontaxonomic indicators of eutrophication should be studied further.

In addition to identifying species that provide good indications of changes in the intensity of different classes of stressors (e.g. organochlorines vs phosphorus), a second objective in selecting a suite of indicator species should be to identify species that utilize different habitats within the ecosystem and, thus, may provide habitat-specific indications of stressor exposure and effects. For example, bottom-feeding fish (e.g. the brown bullhead and the white sucker) can provide information on stressor intensity in benthic habitats (e.g. contaminated sediments) that would be complementary to information gathered from species such as lake trout, that are pelagic for most of their life cycle. Analysis of bottom-feeding fish may be especially important since sediments represent a sink for toxic substances entering the lakes

(Delfino 1979). Many other examples of habitat partitioning among species (e.g. phytoplankton vs periphytic algae) exist for commonly identified groups of indicator organisms.

Diagnostic indicators have been used to implicate specific classes of chemicals and other stressors in observed environmental impacts. A suite of diagnostic indicators, including biochemical and physiological measurements as well as field experiments (e.g. egg-swapping between affected and unaffected colonies), have been used to isolate the cause of observed effects at the individual and population levels in fish-eating birds (e.g. Gilbertson 1988; Peakall 1988). Similar diagnostic tools are available for other indicator organisms, for example measurements included in the dichotomous key for the lake trout (Ryder and Edwards, 1985) provide a degree of diagnostic capability. These tools should be developed for other indicator organisms, such as bottom-feeding fish species, so that, in the event that objectives regarding the removal of stressor effects (e.g. elevated tumor incidence) are not achieved as a result of current remediation efforts, the cause can be isolated and modifications in management and restoration programs can be effected.

Early warning indicators are available for the measurement of several of the candidate indicator organisms discussed. Nonspecific, sublethal early warning indicators serve a dual purpose as sensitive companion indicators to compliance specific indicators: 1) to signal impending dete-

rioration in environmental conditions and 2) to judge the need for continued remediation efforts after compliance indicators (e.g. population abundance) have achieved a level stated by an ecosystem objective. For example, the number of breeding pairs of bald eagles may recover to acceptable levels but eggshell thinning may continue to be detected.

4.3.4 Measurements on Communities and Ecosystems

Historically, environmental monitoring efforts have focused on the identification and use of indicator species to detect ecosystem deterioration. As already discussed, valuable information can be obtained through the measurement of various biochemical, physiological, organismal and population parameters as to the effects of specific (e.g. organochlorine pesticides) and more general forms of ecosystem degradation in the Great Lakes region. However, several limitations occur in any program that relies solely on indicator species for monitoring environmental change:

The limited geographical and/or temporal distribution of many species limits their usefulness as environmental monitors to restricted areas in the Great Lakes region, while communities of organisms are ubiquitous throughout the region and through time. Transferability of many community/ecosystem parameters across ecosystem boundaries facilitates their use as indicators that express changes at regional scales.

Ecosystems typically exhibit a high degree of functional redundancy, such that effects observed on a few species do not necessarily translate into impacts on ecosystem operation (Hill and Wiegert, 1980). Thus, while selected species previously targeted as indicator organisms serve several useful purposes in the context of a comprehensive monitoring program, these indicators used by themselves may not be adequate for accurately assessing ecosystem integrity.

Measurements performed on communities and ecosystems consider the dynamics and responses of many constituent populations, facilitating more robust environmental monitoring, and should reduce the frequency of false negatives and positives, if properly applied.

Although societal concern over environmental degradation has traditionally focused on selected species of commercial or aesthetic importance, public concern over broader environmental issues (e.g. maintenance of biodiversity) is increasing.

A community/ecosystem approach to environmental monitoring is not without drawbacks. The very properties that enhance the robustness of these measures (i.e. the

incorporation of many effects at the population and lower levels) inevitably reduce their diagnostic and, in some cases, early warning potential. Community and ecosystem responses to environmental stressors are complex and, in most cases, not well understood. Thus, advancement simultaneously toward specific objectives of ecosystem restoration projects in the short-term and broad goals related to a self-maintaining ecosystem in the long-term will undoubtedly involve simultaneous monitoring at many levels of biological organization to provide a clear picture of ecosystem condition.

Selection of indicator populations and communities may be best viewed as complementary, rather than competing, tasks. Indicator species appear to be most effective at: 1) directly measuring progress towards the restoration and maintenance of populations that possess commercial and/or social value and 2) tracking progress towards remediation of specific forms of environmental impact by identifying species known to be especially sensitive to individual stressors. In contrast, a community/ecosystem level approach to environmental monitoring provides a more robust assessment of ecosystem health in the Great Lakes as it is affected by the cumulative effects of many stressors, ranging from persistent contaminants to the introduction of exotic species.

Community Structure

The ecological community can be broadly defined as all the species present in a given habitat that have the potential to interact in some manner. However, the term "community" is generally operationalized so as to encompass only those species of a particular taxonomic group of interest to the observer. This is not to say that taxonomically dissimilar species do not interact strongly (e.g. fish and zooplankton) but, rather, that available data are largely dictated by taxonomic considerations. Reasonably well studied "communities" in the lakes region include fish, bird, zooplankton, zoobenthic, meiofauna, phytoplankton and periphyton.

Numerous measures of community structure have been used as indicators of the response of natural ecosystems to anthropogenic stress, and no single measure enjoys unequivocal support as a consistently superior measure of ecosystem integrity. The seemingly vast array of available parameters used to define community structure generally encompasses a limited number of structural attributes:

- Number of species
- Relative abundance/dominance
- Biomass
- Guild structure
- Size spectra
- Foodweb (trophic) structure

The most basic parameter defining community structure is that of species diversity. This measure considers both the number of species present in a community (species richness) and their relative abundance (species evenness). Measurement of species diversity is one of the most commonly-used parameters for assessing an environmental condition. The ability to summarize information on species richness and evenness in a single value, using a diversity index, undoubtedly explains part of the appeal of these measures. Purported theoretical relationships between diversity and stability have led to a widespread belief that high species diversity is a property of healthy (i.e. stable) ecosystems and that decreases in diversity signal environmental deterioration and the loss of ecosystem integrity (Pontasch et al. 1989).

In practice, several problems may limit the usefulness of routine diversity measures as indicators of ecosystem deterioration and recovery. A number of diversity indices have been proposed and none has logical or practical primacy. Older indices (e.g. Shannon's diversity; Margalef 1958) continue to be used uncritically, despite several recognized drawbacks (e.g. Hurlbert 1971). A lack of clear selection criteria is problematic since different measures can be given substantially different indications of trends in diversity among communities at different locations (Hurlbert 1971). Empirical evidence does not support a consistent relationship between diversity and environmental stress (Connell 1978; Ward and Stanford, 1983; Stevenson 1984). Thus, while changes in a diversity measure per se may provide an indication of changes in environmental quality, the measure provides no indication of whether conditions are improving or deteriorating. As elsewhere, diversity measures have been widely used as measures of community response to pollution, both in the Great Lakes region and elsewhere. Results of laboratory and field studies (reviewed in Fitchko 1986) repeatedly show that changes in diversity do not provide reliable indications of changes in the degree of ecosystem impact.

Many other measures for summarizing species evenness (i.e. equitability indices) are also available, and their relative performance can differ greatly (e.g. Molinari 1989). As with diversity indices, often no clear trend is found for this response in relation to toxic stress (Weber 1973; Kraft and Sypniewski, 1981).

Consideration of species richness alone generally provides a more accurate assessment of stress-related changes in natural communities than the use of diversity indices (Patrick 1963; Cairns et al. 1988). Measurement of species richness is complicated for several reasons. It is inherently difficult to detect the presence of rare species for obvious reasons. Patrick and colleagues (1954), for example, estimated that a minimum of 8,000 algal cells must be counted to accurately estimate the species richness of stream diatom

communities; the time and taxonomic expertise required to obtain such information is, therefore, very great relative to many other measures. Indeed, the taxonomy of many groups is not well known or requires laborious methods. Thus, although species richness is a potentially powerful measure of environmental stress, the difficulties associated with its measurement make it impractical as a routinely-monitored environmental indicator.

Data collection requirements for the calculation of diversity or evenness indices are comparable to those needed to perform analyses of changes in the taxonomic composition of the community. Community comparisons based on taxonomic similarity, which consider the identities of species present as well their relative abundances, often provide a more powerful means of detecting patterns of community change than do measures of species diversity (Marshall and Mellinger, 1980; Pontasch et al. 1989). Because the identity of species is ignored in the calculation of diversity indices, these measures are not sensitive to compensatory changes in the community, such as the replacement of one dominant species by another, which alter the taxonomic composition of the community, but have little effect on community diversity. Such problems are corrected by the use of taxonomic similarity methods. Similarity methods are applicable both to the analysis of spatial and temporal trends in community structure.

A second alternative to the diversity approach is a large class of measures that evaluate environmental conditions in terms of the relative abundance of sensitive and tolerant taxa in the community. These biotic indices require detailed information on the autoecologies of individual species and, thus, are generally proposed as a result of intensive study of a particular watershed or region. The extent to which a given biotic index is transferable from one region to another will depend on several factors, including the degree to which species in the community exhibit a cosmopolitan (i.e. global) distribution. This condition may explain the preponderance of these indices for microbial groups such as algae (e.g. Kolkwitz and Marsson, 1908; Pantle and Buck, 1955; Lange-Bertalot 1979; Descy 1979) since these organisms tend to exhibit a more widespread geographical distribution than most larger organisms. Biotic indices summarize the responses (i.e. presence or absence and, often, a semiquantitative measure of abundance) of several indicator species to stress to provide a single number that characterizes ecosystem condition. This approach, therefore, suffers from some of the same limitations inherent in the use of indicator organisms, particularly the fact that the relative tolerance of a species to stress varies with the type of stressor being applied (Cairns et al. 1972; Patrick 1977; Sloof 1983; Mayer and Ellersieck, 1986). Consequently, biotic indices are most feasible in areas that are affected predominantly by one form of stress, a situation which is not common in the Great Lakes. Species have historically been classified with

regard to their tolerance to organic enrichment (Kolkowitz and Marsson, 1908; Beck 1955; Hilsenhoff 1982). Attempts at classifying populations in a community with regard to their tolerance to other types of stress have met with varying degrees of success (e.g. Review in Fitchko 1986). At present, no one type of index enjoys overwhelming appeal, and most groups currently used in this manner (e.g. oligochaetes) are used in reference to organic enrichment.

The biomass or standing crop of a particular community is an extremely coarse indicator of community changes related to environmental stress, but may be useful in certain instances. For example, the yield of commercially valuable fish from the lakes may, for example, provide a good compliance indicator of success in achieving a self-sustaining fishery. Phytoplankton standing crop (e.g. chlorophyll *a* biomass) is used as a measure for assessing trophic conditions (e.g. Hunsaker and Carpenter, 1990). Biomass estimates alone rarely provide adequate information for assessing environmental conditions, but may be useful as part of an integrated index (e.g. trophic state index).

Analysis of changes in biotic size spectra has been proposed as a means of evaluating ecosystem condition (Kerr and Dickie, 1984). The basis for this measure is the observation that stable ecological assemblages exhibit a constant relationship between the size of individuals and their relative abundance in the community. Because stressors tend to differentially affect larger organisms (Woodwell 1967; Regier 1979), increases in the intensity of stress should be manifested as a shift toward increasing dominance by smaller size classes within the community. The Great Lakes fishery has seen a shift from dominance by larger, long-lived species (e.g. lake trout and walleye) to smaller, short-lived species (e.g. alewife and smelt) (Regier 1979). Large diatoms were found to be more sensitive to PCBs than smaller planktonic algae in marine environments, resulting in a shift in community composition toward smaller size classes (Biggs et al. 1978; O'Connors et al. 1978). Such shifts may destabilize pelagic food chains by favoring shifts in zooplankton community composition (Fisher 1975) and, possibly in turn, the fish community. The zooplankton community in a mesotrophic lake shifted toward increasing dominance by smaller forms when exposed to the insecticide, permethrin, in limnocorrals (Kaushik et al. 1985). Shifts in the size spectra of a community can affect the dynamics of aquatic food webs (O'Connors et al. 1978) and, thus, may provide an important indication of ecosystem stability.



Aerial view of algae along shoreline.

Structural indicators discussed above are applicable to most commonly-studied ecological communities in the Great Lakes region. Available evidence does not unequivocally support the use of any one community as an indicator in all situations. As with the selection of indicator organisms, certain general principles apply when evaluating the usefulness of various communities as indicators of different aspects of ecosystem health. For example, communities that encompass populations with short generation times (e.g. microbial communities) can be expected to respond more rapidly to acute stress than do those with longer generation times (e.g. fish). However, communities containing longer-lived species are generally better at integrating longer term effects of certain types of stress (e.g. persistent contaminants). Communities containing species that fulfill several roles in the ecosystem (e.g. herbivores, predators or scavengers) will likely be better integrators of different forms of stress, while those containing species performing very similar functions (e.g. phytoplankton communities) will generally be more diagnostic of a particular types of stress (e.g. phosphorus loading). Benthic (i.e. attached) communities are generally better indicators of local conditions because of their sedentary nature, while planktonic communities may integrate conditions across larger spatial scales. Stressors vary greatly in their mode of impact on the ecosystem, for example changes in phosphorus availability directly affect the algal and macrophyte communities, while overharvesting affects the fish community. Although these initial impacts create several indirect effects on other organisms in the ecosystem, the time-frame for such effects to become noticeable may be much greater. These considerations illustrate certain inherent limitations to the use of any single taxonomic or functional group of organisms for monitoring environmental conditions.

Fish communities of the Great Lakes have been well studied and have been affected by the cumulative effects of a variety of anthropogenic stressors, including the introduction of exotic species such as parasites (e.g. sea lamprey) and competitors (e.g. salmon [*Onchorhynchus*]), commercial exploitation, eutrophication, toxic contaminants and loss of certain physical habitats (e.g. spawning grounds). The ecological and commercial importance of these communities enhances their usefulness as high profile compliance indicators. Many diagnostic and early warning indicators (e.g. physiological responses to stressors) are available to support assessments at the community level. As with any community, certain attributes of fish assemblages limit their usefulness in some instances. For example, the mobility of species within these communities may reduce their usefulness as indicators of change in local environmental conditions. However, this same attribute is advantageous for basin-wide monitoring since it enhances the ability of these communities to integrate the effects of stressors over large spatial scales. Many species utilize several different habitats within the ecosystem during the course of their life cycles, further enhancing their usefulness for basin-wide monitoring. Given the ecological and social importance attached to these communities and the large amount of information available on their structure, they are certainly an essential part of a comprehensive monitoring program in the lakes.

Different taxonomic groups (e.g. insects) within the benthic invertebrate community, or the zoobenthos, have been proposed as useful indicators of the extent of anthropogenic impact and recovery. These communities are ubiquitous throughout the region and are an important component of benthic food webs, although their social relevance is generally rather low. These communities are typically sedentary and, thus, serve as useful indicators of local conditions, unlike far-ranging taxa, such as fish or plankton. Because of their location at the sediment-water interface, these communities integrate the effects of stressors in both ecosystem compartments. The life cycle of most invertebrates is such that anthropogenic impacts on community structure are integrated over reasonably long time periods (e.g. months). Standard sampling methods are available for adoption, and taxonomic delineations are reasonably stable. Attempts to compile an historical record of changes in this community have been hindered by problems, including historical inconsistencies in sampling methodology and the timing of collections (Barton 1989). Thus, while certain general trends are reasonably well documented (e.g. shifts from a *Hexagenia* "community" to one dominated by oligochaetes), detailed patterns of spatial and temporal change are more difficult to document. Additional problems, many of which are related to the frequency, scheduling and mode of sampling, are described in detail by Barton (1989).

Benthic invertebrates have been used most successfully to evaluate impact associated with organic enrichment. In particular, many observed shifts in community structure have been attributed to the differential ability of component species to tolerate decreases in dissolved oxygen concentrations brought on by eutrophication (Jonasson 1984). While invertebrate assemblages have been shown to be reasonably sensitive to other forms of stress (e.g. heavy metals) in the laboratory, data collected in situ in the Great Lakes and elsewhere do not provide clear evidence to document trends in these communities related to most stressors (e.g. Fitchko 1986). Thus, while the practice of using these communities to monitor the reversal of eutrophication in the lakes (e.g. Reynoldson et al. 1989) appears sound, the extent to which these communities integrate changes in other aspects of ecosystem health remains uncertain.

The phytoplankton communities of the Great Lakes are arguably one of the most underutilized community-level indicators for evaluating long-term ecosystem change and the cumulative success of various restoration efforts ongoing in the region. It is clear that many observed changes in phytoplankton community structure (e.g. species composition) are the direct result of increased inputs of phosphorus into the ecosystems. However, other important environmental changes have been implicated as well, including a gradual increase in conservative ions (e.g. chloride) in the lakes, related to human activities (Stoermer 1978) and, potentially, to changes in the structure of higher trophic levels, such as cascading effects (e.g. Carpenter et al. 1985). Although algae exhibit acute sensitivities to toxic stress within the range of that observed for other organisms (e.g. Patrick et al. 1968), the effects of xenobiotic substances on the structure of phytoplankton communities in the lakes are likely masked by other effects just described.

Analysis of the remains (i.e. silica cell walls) of the siliceous phytoplankton assemblage (i.e. diatoms and chrysophytes) in the sediments of the lakes may provide the best available historical record of presettlement biological conditions and can be used to infer subsequent environmental changes. Paleocological studies have related changes in the taxonomic composition of this community to cultural eutrophication as far back as the early 19th century (Stoermer et al. 1985a, b, 1987; Wolin et al. 1988). These investigations have identified several patterns of community change (e.g. biomass production, species composition) with increasing nutrient enrichment. The historical record may not be the ideal benchmark for gauging the success of ecosystem recovery. The total removal of human sources of impact on the ecosystem is not considered a feasible goal, nor is it considered the optimal goal (e.g. continued harvesting of fish and other commercially important species). Theoretically, it is possible that a stable community structure, quite unlike that which occurred under pristine conditions, may arise as a

result of reduced, albeit continuing, human impacts (May 1977). However, since few suitable reference sites are available to serve as contemporary benchmarks of the "minimally-impacted" state of all the lakes (with the possible exception of Superior), the historical record appears to be the most feasible reference point.

From a practical standpoint, the monitoring of phytoplankton changes may be complicated by the relatively high taxonomic expertise required for detailed community analyses. However, it is quite likely that certain simplifications in counting and analysis may be implemented without much loss of information. Detailed (e.g. annual) assessments of trends in phytoplankton community structure may be rather costly, because of the need to control seasonal and stochastic fluctuations in community parameters in any such analysis. However, long-term (e.g. five year) environmental trends may be monitored in a relatively cost-effective manner, using phytoplankton assemblages.

The potential for other communities in the Great Lakes region to serve as indicators of an ecosystem's condition has not been as thoroughly explored as the three just discussed. The zooplankton community in the lakes, like other communities, has undergone a dramatic change in structure over the past several decades as a result of changes in phosphorus loading and the trophic composition of the fish assemblage (Kitchell et al. 1988). Introduction of the exotic cladoceran, *Bythotrephes*, is forecasted to exert potentially significant changes on this community as well (Scavia et al. 1988). These communities occupy an intermediate position in aquatic food chains and, thus, are strongly affected both by changes in nutrient loading, which alters the quantity and quality of algal resources (bottom-up control), and by changes in the intensity of fish predation (top-down control). Because these two effects, which may result from radically different management strategies (e.g. reducing phosphorus inputs vs implementing fish stocking programs), can elicit similar changes in the zooplankton community, it may be somewhat difficult to attribute shifts in zooplankton community structure to a particular management effort (Gannon and Stemberger, 1978). The effect of toxic substances on the structure of Great Lakes zooplankton communities is still poorly understood (Evans and McNaught, 1988).

Nearshore periphytic (i.e. attached) algal communities may be superior to phytoplankton communities for assessing changes in local environmental conditions related to lake trophic status. Deep-water algal communities may be useful for lakewide monitoring partly because of a lack of strong seasonal successional patterns (Kingston et al. 1983). The utility of using data from zooplankton, periphyton or other communities to indicate changes in ecosystem condition should continue to be reviewed as research proceeds with a focus on the degree to which their use might surpass present monitoring efforts.

Analysis of food web dynamics provides a means of integrating various observed direct and indirect impacts of stressors on different communities into a broader management perspective for the Great Lakes. Food web dynamics are the basis for most aspects of ecosystem operation and, thus, can be directly related to ecosystem integrity (e.g. self-maintenance). The consequences of anthropogenic effects on Great Lakes food webs do not stop at the waterline, as evidenced by impacts on fish-eating birds (e.g. Gilbertson 1988), effects of fish consumption on human health (Jacobson et al. 1984) and economic effects of fluctuations in commercial fish yield. Perhaps most importantly, an understanding of previous, present and possible future trophic interactions seems necessary to help ensure that evolving management programs in the lakes succeed in achieving and maintaining stable ecosystem operation.

Models of food web dynamics are being increasingly used as a predictive tool for ecosystem management. These models are used to forecast the consequences of changes in one biotic compartment (e.g. piscivorous fish) on other organisms in the larger lake community. This type of analysis has several applications to environmental management in the lakes, ranging from the consequences of overharvesting a particular fish species or declines in top predators resulting from contaminants, to the accidental or planned introduction of a new species into the ecosystem (e.g. Cohen 1989; Fontaine and Stewart, 1990). Continued developments in the analysis of food web dynamics facilitate the development and simulation of alternative management strategies for Great Lakes biota and for investigations of linkages between exposure in one ecosystem compartment and potential effects in others.

Community / Ecosystem Function

Ecosystem functional processes (e.g. productivity, decomposition) have been widely advocated as important indicators of ecosystem stability or "homeostasis" (Van Voris et al. 1980; Bormann 1983; Odum 1985; Rapport et al. 1985). Odum (1985), for example, predicts several functional responses to stress that signal imbalance in the ecosystem, including increased maintenance costs (elevated rate of respiration per unit biomass) and an imbalance in the ratio of production to respiration, which should be equal in a stable system. The question as to whether ecosystem structure or function is more sensitive to stress may be answered differently, depending on the ecosystem under study. Studies of forest ecosystems generally concur in their conclusion that functional changes (e.g. increased loss of nutrients and decreased rates of decomposition and primary productivity) provide an earlier indication of the onset of ecosystem stress

than do structural changes (e.g. shifts in species composition). Just the opposite may be true in aquatic ecosystems, where unicellular algae are the dominant primary producers. In his work in the Experimental Lakes Area, Schindler (1987) found changes in ecosystem structure (i.e. algal species composition) to be a much more sensitive indicator of ecosystem stress, in this case increased acidity, than were comparable functional indicators (i.e. ecosystem primary productivity). Ecosystem functions mediated by microbes, in particular, may adapt and recover quite rapidly following the onset of stress. Recovery of algal productivity in marine mesocosms exposed to copper occurred concurrently with a shift in taxonomic composition from copper-sensitive to copper-resistant taxa (Thomas et al. 1977). Bacterial assemblages exhibit conferred resistance to certain toxic substances (Pfister et al. 1970; Szczepanik-van Leeuwen and Penrose, 1983), a situation which may result in a recovery of functional activity following exposure. Thus, the relative performance of structural and functional indicators of stress appear dependent on the ecosystem and organisms under study.

Decisions regarding the use of structural vs functional indicators of an ecosystem's condition must also be based on the goals and objectives of ecosystem management programs. In the case of the lakes themselves, specific structural goals (e.g. fisheries management) are obviously important. While broader objectives, such as the restoration of self-maintaining ecosystems, can generally be framed both in structural and functional terms, little empirical and historical evidence is available for specifying a level of function that indicates achievement of such a goal. In practice, measurements of functional attributes tend to be costly and prone to sampling error (Cairns and Pratt, 1986; Levine 1989). Because any understanding of ecosystem operation in general, and that of the Great Lakes in particular, is still far from complete, it will be quite difficult to develop ecosystem objectives and indicators based on functional attributes of the ecosystem.

An alternative means of monitoring ecosystem processes is to infer changes in functional processes from shifts in community structure. For example, Cairns and Pratt (1986) suggest the analysis of microbial (prokaryote and eukaryote) community composition as an alternative to functional measures, since most of the mass and energy passing through an ecosystem are affected either directly or indirectly by the activity of this biological component.

Landscape Ecology

There is increasing interest in the discipline of landscape ecology and its application to environmental management.

This science seeks to understand relationships between spatial patterns and ecosystem processes. The Great Lakes basin is a complex ecosystem exhibiting environmental heterogeneity over several spatial as well as temporal scales, and it is likely that different processes, including various anthropogenic impacts, are important at each scale. Methodologies under development to study landscape mosaics and identify important spatial scales (i.e. patches) may increase the understanding of how this system operates and responds to anthropogenic stress (e.g. Wessman 1990).

Although the discipline of landscape ecology is still in its infancy, certain developing principles may apply to ecosystem monitoring and management. Karr (1991), for example, proposed measures of patch geometry, habitat fragmentation, and linkages among patches as candidate indicators. Changes in the size, arrangement and isolation of ecologically important habitats (e.g. spawning grounds or wetlands) can have important effects on population and ecosystem processes. Changes in the geometry of adjacent habitats will affect the flow of organisms and material across their boundaries and alter the amount of edge habitat (i.e. ecotones) available for various species. The relationship between such analyses and policy issues such as regional and local development are obvious. Thus, while continued research will be necessary to develop this discipline for practical use, future contributions to ecosystem management seem imminent.

4.3.5 Integrated Measures of Ecosystem Health

It should be clear from the above discussion that several candidate indicators of ecosystem health exist at different levels of biological organization for gauging the recovery and maintenance of ecosystem health in the Great Lakes. Evaluation of the relative performance of different indicators shows that no single measure is consistently superior to all others. In light of inevitable limitations on the use of any single indicator for monitoring ecosystem conditions, various attempts have been made to combine a suite of biological indicators into a robust index of ecosystem health or integrity. The use of an integrated measure of an ecosystem's condition is advantageous since deficiencies in the indicator ability of any one parameter should not invalidate the overall assessment. Development of an index that is sensitive to several different types of stressors would be beneficial because of the complexity of environmental impacts in the lakes region. Integrated indices that reduce information from several measures into a single value are advantageous from a decision-making standpoint, although assumptions involved in this approach (e.g. weighing the importance of individual measures in the index) are problematic (Friend and Rapport, 1990). Other integrated measures of an ecosystem's condition do not attempt data reduction, but

utilize the same criteria for selecting a suite of measures for simultaneous consideration. In particular, measures are generally chosen that provide complementary information regarding environmental effects and are not redundant in terms of information content.

Indices of biotic integrity (IBI) are increasingly being employed to assess and monitor ecosystem health in the United States; these indices use fish and macroinvertebrate communities (Karr et al. 1986; Davis and Lubin, 1989; Plafkin et al. 1989). Several parameters, referred to as metrics, are selected; they reflect individual, population, community and ecosystem attributes. Three basic types of metrics have generally been used to assess ecosystem health by means of fish communities; these types include species richness and composition comprising indicator taxa, trophic composition (proportion of species in different feeding groups) and the overall abundance and condition (e.g. proportion diseased or with tumors). Macroinvertebrate IBIs are typically based on taxon richness of various orders, proportional abundance of various taxa, percentage of "tolerant" species in the community and the dominance of different feeding groups (e.g. shredders vs filter-feeders) in the community. These categories do not encompass all classes of candidate biological indicators discussed in this report, and an index of this type for the Great Lakes might include additional parameters and exclude some of those just listed.

Several candidate indicators may be chosen for initial analysis at selected locations, based on previous information regarding their relationship to unimpaired ecosystem operation and their sensitivity and response to environmental stress. Redundancy analysis (Kaesler et al. 1974) can then be used to assess the extent to which each candidate indicator provides unique information about changes in an ecosystem's condition. These results can be used to reduce the number of indicators in the final index. IBIs constructed for one region usually require modification for use elsewhere because of differences in species composition and ecosystem functional attributes.

The final suite of metrics (i.e. indicators) is used to monitor changes in ecosystem condition. Individual metrics are scored, based on the degree of similarity between values measured at a site of interest and that for some nominal state. Usually, the nominal state is defined as the present-day, minimally-affected condition (i.e. reference sites), although historical information or quantitative ecosystem objectives (e.g. proportional or absolute abundance of *Hexagenia* in the benthic assemblage) could conceivably be used. For example, a particular metric (e.g. total species richness) may be assigned a score of one, three or five points, corresponding to an increasingly "healthy" condition relative to some reference measure. Metric scores are summed to provide a single number that, when compared to the desired condition, rep-

resents the degree of impact at a particular location.

Indices of biotic integrity have been primarily applied to streams and rivers, although the concept is certainly applicable to other ecosystems as well. Because that each of the lakes and the St. Lawrence River are unique in terms of their biota, indices of this type would have to be tailored for each of these ecosystems.

The IBI approach is not without certain limitations. Steedman and Regier (1990) note that, in aggregated form, the index has little or no diagnostic value and lacks any theoretical underpinning. However, they also note that individual indicator parameters that comprise such an index do relate to important ecosystem properties and can provide insight as to the cause of ecosystem stress. In terms of the framework proposed in this report, measures incorporated into an IBI would be supported by more detailed diagnostic indicators to aid in identifying the cause of impact. Concern has also been raised over the lack of an objective means for weighting the various parameters used to construct an index such as the IBI (Friend and Rapport, 1990). In constructing a macroinvertebrate IBI (the Invertebrate Community Index) for Ohio, Davis and Lubin (1989) used principal components analysis as a statistical means of providing weightings for individual metrics.

The Ecosystem Distress Syndrome (EDS) (Rapport et al. 1985) defines ecosystem stress in terms of a host of parameters, including changes in community size-spectra, species richness, species composition to favor opportunistic or tolerant species, the incidence of disease, population stability (e.g. "blooms" or outbreaks) and the degree of bioaccumulation of contaminants. As with the IBI, several indicators are selected that together are capable of providing a robust assessment of ecosystem condition, again in reference to some desirable state. Unlike the IBI, the EDS makes no attempt to construct an agglomerative index from these various measures.

The harmonic community concept developed by Ryder and Kerr (1978) offers an integrated approach to the use of fish communities for assessing ecosystem integrity in the lakes. This approach views the fish community (and, indeed, the larger lake community) as an evolutionary entity structured by a variety of species interactions of varying complexity and strength (e.g. parasitism, niche separation). Several properties of harmonic communities have been identified (Ryder and Kerr, 1990) and operationalized in terms of measurable parameters, such as niche complementarity among component species, population dynamics, particle-size density and specific trophic interactions. These authors note, however, that, as with the ecosystem distress syndrome just discussed, component parameters have yet to be synthesized in the form of a single "index of ecosystem integrity" in a

manner similar to the index of biotic integrity.

Ultimately, the successful development of integrated measures of ecosystem health will depend on the elucidation of measurable properties related to ecosystem integrity. Recent efforts toward such a system with respect to the Great Lakes (e.g. Edwards and Regier, 1990) are notable and should serve as a foundation for future advancements.

4.4 *Socioeconomic Indicators*

4.4.1 Introduction

The connections between the condition of the natural environment and human well-being have become less immediate and obvious in the past century. The people of the Great Lakes region no longer experience mass mortality from typhoid or starvation from crop or catch failures. In the absence of these gross reminders that people are a part of, and dependent on their natural environments, we are left with more subtle evidence of our connections. Local restrictions on fish consumption or swimming; questions about long-term health effects from contaminated food, air or water; daily frustration with the appearance and odor of the local environment and questions about the sustainability of industries harvesting natural resources all remain. From the many indicators developed to follow social and economic trends, a few are closely linked to the state of the natural environment and will be sensitive to degradation in that part of the environment. These indicators of linkage between humans and the non-human components of their environment can assess not only the effects of human activity on the environment, but also the effects of environmental degradation on human well-being. They provide evidence for the social and political relevance of ecosystem objectives that lack a human face. By documenting the linkages between ecosystem health and human well-being, the societal will to protect ecosystem health, despite costs, can be reinforced. Socioeconomic indicators provide information useful to policy-makers, e.g. are we meeting specific management goals for the sustainability and protection of human health and well-being; what are the risks and financial and environmental values of different human activities; what is the contribution of the natural environment to regional wealth?

Responses that link the socioeconomic health of the Great Lakes basin to ecosystem health can be divided into three broad categories. Environmental quality must be sufficient to maintain:

- **Human health**
- **Reasonable human uses of resources**
- **Favorable public perception of the quality of life and the environment**

We focus here on responses to environmental degradation (impacts) and not on documentation of the sources of that degradation (inputs), described in the physicochemical section of this report. When any indicator used for assessment or trend monitoring, either biological or socioeconomic, suggests that conditions are unacceptable, diagnosis must follow (Figure 1). Correlations between impact indicators, exposure indicators and input indicators will provide the evidence for linkage between the environmental degradation and impairment of socioeconomic functions.

4.4.2 Human Health

To the great majority of people, the protection of human health is the most important goal of environmental management. There is no goal with higher social relevance. Polls have shown that people are unwilling to accept even minimal additional risks to human health as a consequence of environmental degradation from industrial activity, and the majority of people profess a willingness to pay more for products in order to reduce such risks (Gallup 1990; Harris 1990; Bird and Rapport, 1986).

Even with this apparent commitment to the goal of preventing human health effects from environmental degradation, there are serious problems in designing an effective program to monitor human health effects. Gross effects of pollution are always much easier to detect than subtle, rare or long-term effects. Fortunately, gross effects on human health are rare in these times. It is the subtle, rare or long-term effects that must be monitored, and it is difficult to detect these effects with certainty. Study designs and possible endpoints are outlined in Table 2. Study designs vary in social relevance and interpretability, with epidemiological studies being most relevant but least interpretable, and studies with surrogate species most interpretable but least relevant. Categories of indicators that can be monitored in any study design cover all organ systems and all stages of disease progression. Indicators of impact range from the most relevant measures of fully-developed disease to quicker, cellular or behavioral measures of stress, that may be useful as early warnings. The spatial scale of the study will be dictated by the probable route of exposure. Exposure through drinking water dictates a local spatial scale. Exposure through consumption of open water fish dictates a lakewide spatial scale.

Several IJC reports have addressed the factors contributing to uncertainty in determining human health effects of a degraded environment (IJC 1986; IJC 1990; Colborn 1990). Uncertainty results from the ethical imperative that studies with humans are correlative, not experimental. Studies with humans typically encompass multiple causative agents, not all related to the environment (e.g. adults in a study population sometimes smoke cigarettes). So conclusions about causality linking environmental agents to human health effects are weakened. To address this concern, experiments designed with surrogate species can be used, but the direct applicability of animal data to human health consequences is debated (Lave et al. 1988). It is also difficult to detect the long-term effects of environmental degradation on human health in a timely manner, yet warning in time to prevent human health consequences is clearly desirable. In order to provide information in a timely manner, it is necessary to rely on indicators that occur early in the progression of disease, before the fully-developed adverse effect occurs. These indicators are usually small, quick and relatively unimportant in and of themselves. Ames testing of drinking water, reproductive health of feral sentinel animals and physiological biomarkers in exposed human populations are promising early warning indicators of human health effects. But their effectiveness as assessment or compliance indicators depends on the establishment of a clear relationship between these indicators and ones with more obvious biological and social relevance.

Epidemiological studies of exposed human populations provide the most convincing evidence of human health effects. These studies have very high social relevance, but they often lack interpretability, timeliness or generality across stressors. The level of effort, cost and conclusiveness of epidemiological studies varies with design (Marsh and Caplan, 1987). Ecological studies correlate disease incidence (from registries) with general measures of exposure on a gross scale (e.g. Page et al. 1976), and are less costly and conclusive. Retrospective case-control studies are intermediate in cost and more easily interpreted. Cohort studies (e.g. Fein et al. 1984) are more costly and more conclusive, but still fall short of establishing a cause-effect relationship.

There are background data on the human health effects of environmental degradation of the Great Lakes basin, but results are equivocal (eg. Flint 1991; Colborn 1990; IJC 1990; Fitchko 1986). A chemical specific analysis indicates some cause for concern because chemicals that are individually toxic in laboratory tests with surrogate species are detected in the Great Lakes environment. However, there is less direct evidence for human health effects linking observed impairments in human health to existing environmental degradation. Of the three major routes of human exposure, drinking water, fish consumption and aerosols, fish eating is generally thought to present the greatest exposure and risk. The most

direct evidence for adverse human health effects from environmental pollution is found in a series of studies linking PCB exposure through consumption of contaminated fish to human health effects. Infants of mothers consuming fish from the Great Lakes were smaller than controls (Fein et al. 1984). Such infants also had behavior deficits (Jacobsen et al. 1984) and impaired visual recognition, an indicator related to future intellectual functioning (Jacobsen and Jacobsen, 1988). However, no adverse health effects were clearly related to PCB exposure in fish-eating adults (Humphrey 1988). Replicating and continuing these types of epidemiological studies provides the most relevant and convincing evidence of the status of human health. However, to be used as a monitor of environmental condition, such studies would need to maintain a broad focus in order to include possible effects from stressors other than PCBs and from new stressors. There is some evidence that cognitive function in infants is sensitive to a range of toxic substances and may be a general indicator (Jacobsen and Jacobsen, 1988). An early warning capacity is also desirable and may be found in the use of biomarkers, subclinical indications of a future adverse effect.

In contrast to the paucity of direct evidence of adverse effects on human health, there is an abundance of evidence relating health effects on feral species to environmental degradation in the Great Lakes (Reviews by Colborn 1990; Gilbertson 1988). In addition to their intrinsic value, these species may well be effective sentinels for the assessment of human health effects, similar to the canary in the coal mine. Studies of feral populations have good biological relevance, and some social relevance. Because of differences in the ways that even closely related species respond to the same chemical, there will be uncertainty in predictions of human health effects from observations on sentinel species. However, because studies on sentinel species can be more manipulable than is possible when studying humans, they may be more interpretable. When the life cycle of the organism is short, these studies can be anticipatory for human health effects. They are less costly than laboratory toxicity tests on similar surrogate species. They integrate the effects of simultaneous and sequential exposure to many different pollutants found in the environment. The utility of feral populations as sentinels for human health effects would be improved by studies linking exposure biomarkers (e.g. tissue concentrations or induction of relevant enzymes, such as mixed function oxygenase [MFO]) and health effects in the sentinel species (both subclinical biomarkers, such as sister chromatid exchange, and full blown impairments, such as birth defects) to the same indicators in exposed human populations. Some of these linkages have been made. For example, tissue concentrations of PCBs in feral fish and humans have been compared (Hallett 1986). By establishing these relationships, the social relevance and early warning capability of biomarkers would be vastly improved.

TABLE 2. Potential indicators of the response of human health to environmental degradation

A. STUDY DESIGNS — ASSESSMENT APPROACHES WITH DIFFERENT RECEPTOR ORGANISMS

- | | |
|---|---|
| 1. Epidemiological studies on exposed human populations (see March and Caplan, 1987) | a. Environmental studies
b. Case control studies
c. Cohort studies |
| 2. Studies on sentinel species of exposed feral animals (see Gilbertson 1988; Colborn 1990) | a. mammals; minks, voles
b. birds; herring gulls, Forster's terns, eagles
c. fish; spottail shiners; brown bullheads |
| 3. Studies on surrogate species of exposed laboratory animals (see Lave et al, 1988) | a. mammals; mice, rats
b. nonmammalian systems; tissue culture, bacteria (Ames assays), planaria, hydra, water fleas, frogs, fathead minnows |

B. CATEGORIES OF INDICATORS

- | | |
|---|---|
| 1. Neurotoxicity
(see Caplan and Marsh, 1987) | a. <i>in vivo</i> <ul style="list-style-type: none"> • regional incidence rates for multiple sclerosis, Parkinson's, amyotrophic lateral sclerosis • behavioral assays; infant cognitive function, speech, gait, visual disturbance, headaches, memory function • biomarkers; biopsy and histopathology, visual-evoked response, electroencephalogram positron emission tomography, CAT scan, electromyography |
| | b. <i>in vitro</i> <ul style="list-style-type: none"> • cell culture excitability, synaptic potential, repetitive firing properties, nerve conduction velocity |
| 2. Reproductive toxicity
(see Caplan and Marsh, 1987) | a. <i>in vivo</i> <ul style="list-style-type: none"> • regional incidence rates for birth defects, infertility, miscarriage, stillbirth, low birth weight • biomarkers; sister chromatid exchanges, sperm counts, motility and morphological abnormality |
| 3. Carcinogenicity / Mutagenicity / Genotoxicity
(see Sandu and Lower, 1987; Wang et al, 1987; Colburn 1990; Caplan and Marsh, 1987) | a. <i>in vivo</i> <ul style="list-style-type: none"> • regional incidence rates • biomarkers; DNA adducts, sister chromatid exchange, DNA unwinding, histopathology |
| | b. <i>in vitro</i> <ul style="list-style-type: none"> • histopathology of tissue cultures • Ames mutagenicity tests |
| 4. Cardiovascular disease | a. <i>in vivo</i> <ul style="list-style-type: none"> • regional incidence rates |
| 5. Immunocompetency | a. <i>in vivo</i> <ul style="list-style-type: none"> • blood cell counts |

Most assessments of human health effects of environmental pollution have been made using surrogate species. Commonly, a laboratory test exposes a laboratory population of a surrogate species, such as mice, to a single chemical. Dose-response relationships are determined and used to establish safe concentrations and standards. Because of the many problems in extrapolating from data on response to a single chemical to response to a complex mixtures of chemicals, the biological relevance of such tests is not high (Vouk et al. 1987). Ambient toxicity tests, which test the complex mix-

ture as it occurs in the environment directly, avoid this problem. However, problems with the high cost of tests, and great uncertainty due to variability in response, even across closely related species, remain (Lave et al. 1988). So, despite the wide-spread reliance on surrogate species tests to set standards for protecting human health, the scientific community has doubts about their biological and social relevance, their cost-effectiveness, their sensitivity and their interpretability.

In studies with humans, sentinels and surrogates, there are many indicators of health that can be assessed (Table 2). These indicators vary across organ systems, across disease progression and across levels of the biological hierarchy, from subcellular to whole organism. The whole-organism, fully-developed clinical indicators (e.g. cancer mortalities) are more relevant and interpretable, but less timely than the subcellular biomarkers or bacterial surrogate indicators (mutagenicity tests). It is necessary to firmly establish the relationships between biomarkers and future clinical expression of disease before biomarkers can be considered sufficient evidence for regulatory action.

The monitoring of human health effects resulting from environmental degradation is clearly one in which current scientific methods are not yet adequate for the task mandated by the public will. Because of the importance of the objective to the public, management action may be encouraged, despite considerable uncertainty. But that same uncertainty compromises the legal defensibility of the indicators and increases the likelihood of legal challenges to proposed management actions. Effort devoted to the further development of promising methods is justified.

4.4.3 Reasonable Human Use

In the development of economic theory, water, air, soil, feral plants and animals of good quality have been traditionally thought of as free and inexhaustible, and, as such, without value. Because we have found that the supply can be exhausted and the quality impaired, despite technological advances, the conception of these elements as common environmental capital has replaced the earlier notion. Attempts have been made to incorporate this common environmental capital into existing economic instruments that guide policy-making, e.g. environmental impact analysis, cost-benefit analysis, decision analysis (Jansen 1991; Maguire 1988; OECD 1982; Seneca 1987; Wise 1988). However, because most environmental goods and services are not traded on the open market, there has been great difficulty in assigning values to these elements (Jansen 1991; Nijkamp and Soeteman, 1988; OECD 1989). Monetary approaches persist despite this difficulty because dollars provide a means of weighting disparate elements of the environment and economy in order to link them.

It has been suggested that bringing environmental considerations into existing economic instruments is doomed to fail because this approach retains the traditional view of man as master of the natural world and results in a persistent and entrenched undervaluation of ecological attributes (Stewart 1987). This approach must be replaced with an economic model, based on the premise that humans are a part of the

environment and not separate from it. Integrated environmental-economic models have been developed (e.g. spatial economic-environmental models, environmental evaluation models, input-output models, dynamic stock-flow models and materials balance models), but most focus on inputs rather than impacts and ignore ecological processes. In addition, practical applications on a regional scale are lacking (Nijkamp and Soeteman, 1988).

Progress in establishing the linkages between environmental health and economic consequences is being made on a more limited scale. Work towards a natural resources accounting system (NRA) in Canada (Friend and Rapport, 1990; Bird and Rapport, 1986) and certain methods developed for policy-making techniques, such as decision analysis (Maguire 1988) or environmental impact assessment, provide examples of ways in which biological and socioeconomic data can be integrated. These examples point to several types of economic monitoring which would contribute to environmental quality control and environmental policy decisions. First, because sustainability is a specified goal in and of itself, monitoring stocks, harvesting rate and recruitment rate for environmental goods will allow us to evaluate whether the goal of sustainability is being met. Sustainability would also require that environmental services, including recreation and aesthetics, be maintained. Monitoring success in achieving the goal of sustainability does not require translation into monetary terms. In addition, the interpretability of data is facilitated by the clear statement of the goal. Conditions are unacceptable if stock size is depleted, if harvesting exceeds recruitment or if there is a decline in aesthetics or recreational utility. Second, it would be useful to document the current and future contributions of environmental goods and services to regional wealth. These contributions represent the common environmental capital that is risked through poor stewardship. The translation of environmental goods, services and management costs into monetary terms is necessary in this case. This background data would be useful to decision-makers, but there are many methodological problems.

Major categories of human use that are likely objects of protection are listed in Table 3 with examples of potential indicators. Indicators include monetary estimates, actual market values, shadow prices and contingent valuation (willingness-to-pay or adequate compensation for loss estimates). Non-monetary estimates include stocks and flows in terms of biomass, counts of standard violations or instances of inadequate quality, and human preferences.

The usefulness of these potential indicators vary widely. Common problems include lack of any historical data base, lack of consistency or reliability in valuation methods, lack of consistency in data collection methods by jurisdictions, problems in interpretation of changes due to large variations, and

TABLE 3. Potential indicators of the response of human use to environmental degradation

	QUANTITY	QUALITY	VALUATION COSTS	MANAGEMENT
Commercial Fisheries Bird & Rapport, 1986	<ul style="list-style-type: none"> • stock, harvesting, recruitment estimates 	<ul style="list-style-type: none"> • presence of preferred species • restriction on consumption • incidence of tainting, deformities 	<ul style="list-style-type: none"> • shadow pricing: farm reared vs feral fish • employment and payroll 	<ul style="list-style-type: none"> • stocking • lamprey control
Drinking Water Wentworth et al. 1986	<ul style="list-style-type: none"> • stock, withdrawal, replenishment estimates 	<ul style="list-style-type: none"> • treatment costs • chemical and bacterial standards violations • restrictions on consumption • reported acute illness • user satisfaction* 	<ul style="list-style-type: none"> • contingent valuation: willingness to pay and compensation for damage* 	<ul style="list-style-type: none"> • treatment costs
Recreation Hunsaker & Carpenter, 1990; Lichtkoppler & Hushak, 1989	<ul style="list-style-type: none"> • visit counts: sport fishing, swimming, boating, bird watching, bird hunting • boat registration • marina and beach counts • marina vacancy rates 	<ul style="list-style-type: none"> • incidence of fish consumption restrictions • incidence of contact sport restrictions • incidence of fish deformities or tainting • catch per unit effort 	<ul style="list-style-type: none"> • employment and payroll • marina sales • admission fees • shadow valuation; pool construction vs beach use 	<ul style="list-style-type: none"> • stocking
Industrial, Energy and Agricultural Water Use	<ul style="list-style-type: none"> • stock, withdrawal, replenishment rates 	<ul style="list-style-type: none"> • productivity, crop, livestock losses attributable to water quality problems • costs of pre-use treatment: descaling, defouling 	<ul style="list-style-type: none"> • compensation for loss of use • increased product cost due to degradation 	<ul style="list-style-type: none"> • cost of post-use treatment
Aesthetics	<ul style="list-style-type: none"> • subjective satisfaction • miles of shoreline 	<ul style="list-style-type: none"> • incidence of objectionable odor* • incidence of turbidity • incidence of algal blooms 	<ul style="list-style-type: none"> • shadow valuations: water-view vs inferior real estate • contingent valuation; willingness to pay and compensation for loss* 	<ul style="list-style-type: none"> • landscape planning
Transportation Water Use	<ul style="list-style-type: none"> • water levels 		<ul style="list-style-type: none"> • employment and payroll 	<ul style="list-style-type: none"> • income lost due to restrictions on dredging • costs of disposal for contaminated dredge spoils • costs of pollution controls • costs of control of nuisance growths: macrophytes, zebra mussels
Human Health	<ul style="list-style-type: none"> • community level • native people 	<ul style="list-style-type: none"> • perception of a healthy healthy environment 	<ul style="list-style-type: none"> • human welfare • social values 	<ul style="list-style-type: none"> • medical costs • loss of human potential

Support of General Economic Well-being of Region

- traditional economic indicators (GNP, unemployment, income class distribution, etc.)

Future Use

- genetic poll for pharmaceuticals, genetic engineering, temperature buffer in global warming

* Subjective evaluations, dependent on survey of shareholders

lack of logical standards of acceptability. It is likely that various categories of use will be redundant. For example, commercial and recreational fisheries are likely to improve or decline together. As the relationships between indicators is established, redundancy could be eliminated. It is also likely that certain categories of use will be relatively insensitive to changes in environmental quality. For example, waterborne transportation may not be greatly affected by small changes in environmental quality. And indicators of the general economic well-being of the region will probably be largely determined by factors other than environmental health, and, as such, will be relatively insensitive to environmental degradation until it is severe.

Information on human use may be collected at several spatial scales. Drinking water monitoring must be local, as are aesthetics and most recreational uses. But the use of commercial fisheries is information more appropriate for monitoring a larger area. There are few use indicators that are diagnostic or anticipatory. As always, trend monitoring can provide early warning by identifying small but consistent downward trends.

Initial attempts to monitor the sustainability of human use may rely heavily on the best existing data bases and avoid the problems inherent in the valuation of environmental assets until methods are more reliable. Information on commercial fisheries is available (Bird and Rapport, 1986). Data on lake trout and walleye populations provide direct overlap between biological and socioeconomic indicators. These data will track the utility of open-water quality. Data on recreational use are also available (U. S. Department of Interior 1989, as cited by Hunsaker and Carpenter, 1990; Lichtkoppler and Hushak, 1989) and will track nearshore water quality. Additional use categories will require new efforts toward quantification, but, eventually, an inclusive accounting of natural resources in all use categories would be desirable. Methodological problems with valuation methods must be resolved if the overall contribution of environmental goods and services to regional wealth is to be monitored. Studies to determine correlations between different valuation methods and rankings of the acceptance of values by shareholders and decision makers would help clarify acceptable methods.

4.4.4 Perceptions of Environmental Quality and Quality of Life

It is important to distinguish between the assessment of environmental condition, which is an objective process, and the assessment of environmental quality, which is the subjective perception of the adequacy of the environment by an individual. It is the perception of environmental quality which will determine the shareholder's satisfaction with

management efforts. Bird and Rapport (1986) point out two ways that perceptions can then affect the quality of the environment. Individuals can join together to exert pressure that will change environmental policy in both public and private sectors. And individual actions (e.g. recycling, carpooling, source reduction) are an important part of environmental stewardship. To the extent that these attitudes are communicable, an ethic-supporting sustainability is fostered.

Periodic interviews with shareholders can provide data useful in several areas of policy-making and management. First, overall satisfaction with environmental quality can be assessed and used to provide feedback on the success of environmental management and its importance to the shareholder's quality of life (Milbrath 1978). Second, perceptions of the importance of environmental goods and services are largely subjective. Monetary valuation of these assets for complete natural resources accounting, as described in the section on human use, can be estimated from shareholder interviews to determine willingness to pay or compensation for damage (Jansen 1991). Relative importance may change in response to perceived environmental quality and economic well-being, recent publicity about environmental disasters and other factors. Similar non-monetary rankings of aesthetic and ecological values must also rely on public input (Maguire 1988). Third, acceptance of risk from environmental degradation may vary from one impact to the next and among groups of people. Policy-making tools often ignore differences in risk aversion. Interviews can assess which categories of impacts the shareholders are willing to risk for economic benefit and which risks are unacceptable, regardless of benefit. Fourth, interviews of shareholders also serve as a measure of effectiveness of communication on environmental issues. Interviews can determine shareholder awareness of environmental problems, the source of their information and their awareness of available forums for participating in management decisions (Bird and Rapport, 1986). Fifth, level of participation in environmental protection activities can also be assessed, e.g. membership in sporting or conservation groups, attendance at policy-making forums, energy conservation, carpooling and recycling (Gallup 1990).

Although interviews of shareholders and a focus on subjective well-being have been used in social impact assessment for environmental impact analysis and in cross-cultural comparisons, they have not been used as a monitoring or assessment tool. Questions on attitudes to the environment are included in polls by Harris, Gallup and the Centre de Recherche sur l'Opinion Publique, but because there is no consistency in phrasing or in the order of questions, these data cannot be used to assess trends over time (Bird and Rapport, 1986). A standardized instrument is necessary for

monitoring purposes. An instrument for determining perceived environmental quality and subjective well-being has been devised and tested in the Great Lakes basin (Milbrath 1978). This example of an instrument for measuring perceptions of environmental quality includes assessments of the satisfaction and importance of both man-made and natural environmental attributes, focusing particularly on water quality. It has a community level focus, the level at which environmental concerns have the most importance (Bird and Rapport, 1986). A similar instrument, specifically designed for monitoring trends over time rather than differences between populations, and focusing on the environmental quality of the Great Lakes region could be devised (Eyles 1990; Craik and Zube, 1976). Background data would be required to develop and validate the instrument and to define variability in response over time before it would be useful for monitoring trends.

The social relevance of perceived environmental quality and quality of life is unquestionably high. Perceptions of the environment will be sensitive to factors other than environmental change, such as economic well-being, and may be insensitive to some types of environmental degradation. But because sustaining shareholder satisfaction in and of itself is important, the data collected can be interpreted and used to improve management efforts.

Linking subjective perceptions of environmental quality to the objective determinations of scientists is an important subsequent step. When the objective and subjective assessments of environmental quality agree that environmental quality is sufficient, management techniques are vindicated. If objective and subjective assessments reach different conclusions, action is required. More effectively communicating problems and risks to the general public, reformulating goals more in line with shareholder interest, or reordering priorities for addressing existing problems may be appropriate.

There is a corollary benefit to the acquisition of this data. In the process of responding to the interview, shareholders become participants in the process. The act of interviewing constitutes an outreach activity and can promote education about problems and effect simple changes in behavior. Interviews can foster stewardship in this way.

The first part of the paper discusses the concept of development and its measurement. It argues that development is a multi-dimensional concept that includes economic growth, social progress, and environmental sustainability. The paper then reviews the literature on the measurement of development, highlighting the limitations of traditional indicators like GDP and the need for more comprehensive measures.

The second part of the paper focuses on the measurement of development. It discusses the challenges of measuring development and the need for a balanced set of indicators. The paper then presents a framework for measuring development, which includes economic, social, and environmental indicators.

The third part of the paper discusses the implications of the measurement of development. It argues that the measurement of development is not just a technical exercise, but a political one. The choice of indicators and the way they are weighted can reflect different values and priorities. The paper then discusses the implications of the measurement of development for policy-making and for the development process itself.

3.1.4 Perception of Governmental Quality and Quality of Life

The paper reports the findings from the survey on the perception of governmental quality and quality of life. It shows that there is a strong positive correlation between the two variables. The paper then discusses the implications of these findings for policy-making and for the development process.

The paper then discusses the implications of the measurement of development for policy-making and for the development process itself. It argues that the measurement of development is not just a technical exercise, but a political one. The choice of indicators and the way they are weighted can reflect different values and priorities. The paper then discusses the implications of the measurement of development for policy-making and for the development process itself.

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Conclusions and Recommendations

This report has adopted a broad perspective for developing indicators of ecosystem health in the Great Lakes region, that can be incorporated into developing monitoring programs. The continued evolution of monitoring programs in the Great Lakes should be anticipated for several reasons:

- **Relative concern over various types of human impact will change as current restoration activities succeed in their goals and new forms of impact are identified and quantified**
- **Results of continued basic research and surveillance programs will undoubtedly modify the suite of parameters deemed most useful for evaluating ecosystem health**
- **Ecosystem goals and objectives will continue to be developed and refined to meet the broad and changing demands and expectations of various shareholders**

In order to preserve continuity in the face of inevitable change, a comprehensive framework for developing and implementing ecosystem indicators, such as that proposed here, should be adopted for the Great Lakes region.

Based on analyses contained in this report, we provide several general and specific recommendations regarding future directions of environmental monitoring and indicator development in the Great Lakes:

- ***Formulation of Broad Policy and Management Goals and Explicit Ecosystem Objectives.***

Ecosystem monitoring is a costly, yet necessary, proposition. The effectiveness of monitoring programs based on a suite of ecosystem indicators is contingent upon the development of specific ecosystem objectives for ecosystem restoration and maintenance. Restated, it is essential that one knows what he is trying to protect before he can protect it. Identification of the most appropriate ecosystem indicators for various purposes will ultimately be determined by the types of objectives formulated.

Few explicit management objectives have been proposed for the lakes, although those that have been posed are instructive (GLWQA 1987; Bertram and Reynoldson, 1990). While individual ecosystem objectives must be precise in terms of the goal to be achieved, the overall policy and management goals must be broad enough to consider the interdependence among humans and the rest of the environment. Thus, regional goals must consider not only the health of ecological systems, but the health, welfare and activities (both detrimental and constructive) of humans as well.

- ***Integration of Environmental Restoration and Protection.***

Those charged with ecosystem management in the lakes region currently face the principal task of implementing restoration and rehabilitation efforts at various spatial and temporal scales. Because of the number of problems already identified in the region, the necessity of establishing monitoring programs around such efforts is obvious. However, other problems have undoubtedly gone unrecognized or may be impending. Therefore, monitoring programs must also be designed to identify emerging problems in the region and to suggest preventive measures to avert future impacts. Long-term protection of environmental resources will be best assured by the implementation of pre-emptive as well as reactive management strategies.

The framework proposed here allows for the incorporation of reactive and predictive management strategies into an integrated monitoring program. Three basic types of indicators are required to perform the various functions required of a comprehensive monitoring program:

- Compliance indicators, those measurements that can be used to judge whether a stated ecosystem objective has been achieved
- Diagnostic indicators, those measurements that can be used to determine the cause of impacts that prevent the achievement of stated objectives
- Early warning indicators, measurements that are especially sensitive to ecosystem stress and, thus, are capable

of detecting the onset of deleterious conditions before significant impact has occurred. Suitable indicators of each type should be identified to support efforts not only to **achieve** (i.e. remediation) but, subsequently, to **protect** (i.e. prevention) conditions stated in each ecosystem objective.

- ***Development of a Suite of Indicator Species.***

Measurements performed on individuals and populations of indigenous species in the Great Lakes region have proven their utility for indicating the nature and extent of anthropogenic stress on natural biological systems. However, no one species appears to adequately integrate the effects of all important stressors. Certain indicator species can be used most effectively to track progress toward mitigation of the impact of specific stressors on the environment.

While previously-proposed "integrator" species, such as the lake trout and the walleye, appear suitable for use as monitoring tools in the Great Lakes, it is unlikely that these species alone will provide sufficient information to precisely track progress in all aspects of ecosystem rehabilitation. While the "integrator species" approach is appealing, it is problematic for several reasons, particularly because individual species tend to be differentially sensitive to different types of environmental stressors. Previous researchers have identified other species that are known to be especially sensitive to specific classes of stressors. These species represent useful environmental monitors to augment integrator species. Recommended species include the herring gull as an indicator species for monitoring the effects of persistent organic toxicants and an alga, such as *Cladophora*, for monitoring changes in phosphorus availability. Other indicator species of this type would be valuable.

- ***Development of Community- and Ecosystem-Level Indicators.***

There are inherent limitations to the indicator species approach to environmental monitoring. Most indigenous species in the lakes region exhibit a limited geographical distribution and, because they are selectively sensitive to one or a few stressors, may not be suitable for monitoring changes in cumulative impact. Furthermore, difficulties may exist in relating changes in population parameters to the achievement of broader objectives, such as that of a self-maintaining ecosystem. Assessments of ecosystem change in response to ongoing remediation efforts and continued impact in the Great Lakes region will likely be enhanced by the identification of candidate community and ecosystem indicators. Continued basic research in this area is especially important as are interactions between research scientists and ecosystem

managers, aimed at translating concepts related to ecosystem integrity into concrete measures that can be used as indicators of integrity.

Community and ecosystem measurements should complement measures performed on indicator species. While the indicator species approach is suitable for monitoring target populations and the level of impact of specific forms of stress, communities and ecosystems appear more suitable for measuring changes in long-term cumulative impacts. It may be sufficient to perform community and ecosystem analyses at less frequent intervals (e.g. every few years) than those required for population measurements, in light of differences in the focus of these two types of monitoring.

- ***Indicators of Human Health and Linkages to Environmental Degradation.***

Ecological studies with data from registries provide the least expensive way to monitor human health effects over time. Indicators should include cause of death, tumors and birth defects. These studies provide a broadly-applicable and cost-effective approach, but they lack sensitivity and early warning capability and so additional monitoring approaches are necessary.

Fish eating populations represent the most heavily exposed human group. Cohort studies with fish eaters should continue to confirm early indications of adverse effects, to follow-up on groups that are apparently affected, and to monitor these worst-case human health consequences of environmental pollution. Indicators of cognitive function in infants appear sensitive and broadly applicable, but other groups of indicators should be developed.

There are several linkages that can be made to improve the relevance and efficiency of assessments of the effects of environmental degradation on human health. First, physiological biomarkers in exposed human populations are promising early warning indicators of human health effects. But their effectiveness as assessment or compliance indicators depends on the establishment of a clear relationship between these indicators and ones with more obvious biological and social relevance (subsequent full-blown disease). Second, feral animals may well be effective sentinels for the assessment of human health effects, such as the canary in the coal mine. Studies of the utility of feral populations as sentinels for human health effects would be improved by studies linking exposure biomarkers (e.g. tissue concentrations or induction of relevant enzymes, such as MFO) and health effects in the sentinel species (both subclinical biomarkers, such as sister chromatid exchange, and full-blown impairments, such as birth defects) to the same indicators in exposed human populations. Biomarkers that can

be measured in both humans and sentinel species (e.g. herring gulls) should be developed and correlated for future use as early warning systems. Coordination between human and sentinel species researchers should provide the basis for screening biomarkers for usefulness.

- ***Indicators of Reasonable Human Use and Linkages to the Environment.***

Methods for monitoring linkages between the ecological and economic spheres is an area of intense current research, but few suggestions for an integrated regional monitoring schemes have been put forth. The framework described by Friend and Rapport (1990) and the example of implementation found in Bird and Rapport (1986) provide a logical starting point. Stock size, rate of harvesting and rate of recruitment for commercially important fish species is of both economic and ecological importance and avoids methodological problems involved in translating environmental goods and services into monetary terms. The maintenance of recreational utility is included in the concept of sustainability and data is currently available. Eventually, a more complete accounting of human uses should be possible as reliable and standard measurement methods for other categories of use are developed and adopted by localities. Standardized units and methods for natural resources accounting would be of great value.

Methodological problems with valuations must be resolved if the overall contribution of environmental goods and services to regional wealth is to be monitored. Translating these and other unmarketed attributes into monetary terms for compatibility with most economic policy-making instruments requires further research. Studies to determine correlations between different valuation methods and rankings of the acceptance of resulting values by shareholders and decision makers would help clarify acceptable methods. The concepts of value, importance and risk aversion represent linkages between this economic sector and the subjective assessments that are a possible focus of the interviews suggested for monitoring subjective assessments of environmental quality.

- ***Perceptions of Environmental Quality and Quality of Life.***

A standardized instrument should be developed to monitor trends in perceived environmental quality and quality of life for shareholders in the Great Lakes basin. These data will provide feedback on the success of management efforts, determine the relative importance of environmental quality to the shareholders and assess the effectiveness of communication of management problems to the people affected.

Linking subjective perceptions of environmental quality to the objective determinations of scientists is an important subsequent step. When the objective and subjective assessments of environmental quality agree that environmental quality is sufficient, management techniques are vindicated. If both types of assessments agree that environmental quality is lacking, more vigorous management is called for. If objective and subjective assessments reach different conclusions, several possible actions may be called for. More effectively communicating problems and risks to the general public, reformulating goals more in line with shareholder interest or reordering priorities for addressing existing problems may be called appropriate.

6.

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