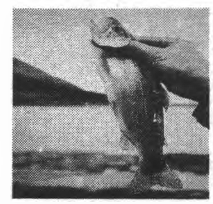
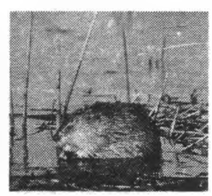
Canada Alberta Northwest Territories

Northern River Basins Study





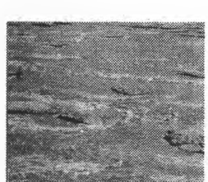



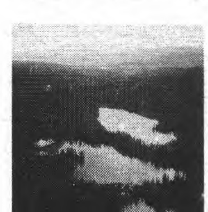


NORTHERN RIVER BASINS STUDY PROJECT REPORT NO. 45

ASSESSING AND MONITORING AQUATIC ECOSYSTEM HEALTH:

APPROACHES USING INDIVIDUAL, POPULATION AND COMMUNITY/ECOSYSTEM MEASUREMENTS




Prepared for the
Northern River Basins Study
under Project 5201-C1

by

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PREFACE:

The Northern River Basins Study was initiated through the "Canada-Alberta-Northwest Territories Agreement Respecting the Peace-Athabasca-Slave River Basin Study, Phase II - Technical Studies" which was signed September 27, 1991. The purpose of the Study is to understand and characterize the cumulative effects of development on the water and aquatic environment of the Study Area by coordinating with existing programs and undertaking appropriate new technical studies.

This publication reports the method and findings of particular work conducted as part of the Northern River Basins Study. As such, the work was governed by a specific terms of reference and is expected to contribute information about the Study Area within the context of the overall study as described by the Study Final Report. This report has been reviewed by the Study Science Advisory Committee in regards to scientific content and has been approved by the Study Board of Directors for public release.

It is explicit in the objectives of the Study to report the results of technical work regularly to the public. This objective is served by distributing project reports to an extensive network of libraries, agencies, organizations and interested individuals and by granting universal permission to reproduce the material.

NORTHERN RIVER BASINS STUDY
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Whereas the above publication is the result of a project conducted under the Northern River Basins Study and the terms of reference for that project are deemed to be fulfilled,
IT IS THEREFORE REQUESTED BY THE STUDY OFFICE THAT;
this publication be subjected to proper and responsible review and be considered for release to the public.

F.J. Wrona — 2 Feb 95

(Dr. F.J. Wrona, Ph.D., Science Director) — (Date)

Whereas it is an explicit term of reference of the Science Advisory Committee "to review, for scientific content, material for publication by the Board",
IT IS HERE ADVISED BY THE SCIENCE ADVISORY COMMITTEE THAT;
this publication has been reviewed for scientific content and that the scientific practices represented in the report are acceptable given the specific purposes of the project and subject to the field conditions encountered.
SUPPLEMENTAL COMMENTARY HAS BEEN ADDED TO THIS PUBLICATION: [ ] Yes [ ] No

P.A. Larkin — 2 Feb 95

(Dr. P. A. Larkin, Ph.D., Chair) — (Date)

Whereas the Study Board is satisfied that this publication has been reviewed for scientific content and for immediate health implications,
IT IS HERE APPROVED BY THE BOARD OF DIRECTORS THAT;
this publication be released to the public, and that this publication be designated for: **[ ] STANDARD AVAILABILITY [ ] EXPANDED AVAILABILITY**

Lucille Partington — 02 Feb. 95

(Lucille Partington, Co-chair) — (Date)

Robert McLeod — 3 Feb. 95

(Robert McLeod, Co-chair) — (Date)

ASSESSING AND MONITORING AQUATIC ECOSYSTEM HEALTH: APPROACHES USING INDIVIDUAL, POPULATION AND COMMUNITY/ECOSYSTEM MEASUREMENTS

STUDY PERSPECTIVE

An important objective of the Northern River Basins Study is to examine the relationships between industrial development and the health of the Peace, Athabasca and Slave River systems. Data obtained over the course of the Study will provide a database that can be used to assess the cumulative effects of man-made developments on the aquatic environment of these rivers. As a part of this exercise, methods and approaches must be identified to measure and monitor aquatic ecosystem health. This study is designed to assist in providing a framework for the development of an ecosystem health/integrity and cumulative effects monitoring program for the northern river basins in Alberta.

> ***Related Study Questions***
>
> *13a) What predictive tools are required to determine the cumulative effects of man-made discharges on the water and aquatic environment?*
>
> *b) What are the cumulative effects of man-made discharges on the water and aquatic environment?*

The traditional approach to setting environmental regulations has been based largely on the assessment of chemical concentrations within receiving environments. The recent movement toward an "ecosystem approach" to environmental assessment and monitoring represents a major shift away from a chemical/physical approach. An ecosystem approach recognizes the complex nature of interactions that occur at a variety of levels of spatial, temporal and organizational scales within the environment. It also acknowledges that human populations constitute an important component of the environment. Although the concept of ecosystem health is difficult to define, it is integral to any application of the ecosystem approach to environmental management.

This project provides a review of the concept of ecosystem health and cumulative effects assessment, and the theoretical framework and practical objectives of these approaches. The report reviews the literature on existing individual, population and community-level approaches and associated measurements being used for the assessment of aquatic ecosystem health and cumulative effects. It also identifies the types of data required to assess and monitor aquatic ecosystem health and cumulative effects in large northern rivers. Finally, this study discusses the applicability of these approaches and associated measurements, and recommends approaches with potential for use in the Northern River Basins Study.

This study recommends that a process is needed whereby scientists and stakeholders begin to develop a well defined system of ecosystem goals and objectives to be used in an ecosystem management program for the Peace, Athabasca and Slave River basins. There is a need for a carefully designed sampling program which will provide the maximum return of information for the time and resources invested. This report recommends the use of life-history end points to assess the general state of populations, and biochemical or physiological indicators to provide early warnings of change and to investigate specific problems. In addition, this study recommends a multivariate approach for community-level biomonitoring within the NRBS, and stresses the need for ongoing experiments at a variety of scales. These techniques are capable of assessing current ecosystem conditions and of providing information on long-term trends within the ecosystem. Any monitoring program should be sufficiently flexible so as to include additional environmental data as they become available.

REPORT SUMMARY

This report has been prepared as part of a larger project whose objective is to develop a framework for the development of an ecosystem integrity and cumulative effects monitoring program for the Peace, Athabasca and Slave river basins. The purpose of this report was to: (1) Review the concepts of ecosystem health and cumulative effects assessment and the underlying framework and practical objectives of these approaches. (2) Critically review the literature on existing population, community and ecosystem-level approaches and associated measurements being used for the assessment of aquatic ecosystem health and cumulative effects and outline the advantages and disadvantages of each. (3) Identify the types of data and information required to adequately assess and monitor aquatic ecosystem health and cumulative effects. (4) Assess the applicability of these approaches and metrics to the Northern River Basins Study and recommend approaches that could potentially be employed to assess and monitor aquatic ecosystem health and cumulative effects within the Northern River Basins.

This study recommends the development of a process whereby scientists and stakeholders begin to develop a well defined suite of ecosystem goals and objectives to be used in an ecosystem management program for the Peace, Athabasca and Slave River basins. There is a need for a carefully designed sampling program which will provide the maximum return of information for the time and resources invested. This report recommends the use of life-history end points to assess the general state of populations, and biochemical or physiological indicators to provide early-warnings of change and to investigate specific problems. In addition, this study recommends a multivariate approach for community-level biomonitoring within the NRBS, and stresses the need for ongoing experiments at a variety of scales. These techniques are capable of assessing current ecosystem condition and of providing information on long-term trends within the ecosystem. Any monitoring program should be sufficiently flexible so as to include additional environmental data and/or improvements in monitoring techniques as they become available.

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1.0 INTRODUCTION

1.1 BACKGROUND

The Northern River Basins Study (NRBS) was established in 1991 and represents a joint agreement between the governments of Canada, Alberta and the Northwest Territories. The study's primary purpose is to examine the relationships between anthropogenic (human generated) development and the health/integrity of the Peace, Athabasca and Slave river basins. More specifically, the NRBS has been mandated with the task of gathering comprehensive information relating to water quality, contaminant distribution and fate, fish and fish habitat, riparian vegetation/wildlife, hydrology/hydraulics and the use of aquatic resources within this region (NRBS 1993). This information will in turn provide a data base that can be potentially used to assess the cumulative effects of anthropogenic development on the water and aquatic environment of the northern river basins.

1.2 GENERAL OBJECTIVES

This report has been prepared as part of a larger project within the NRBS whose objective is to develop a framework for the development of an ecosystem health/integrity and cumulative effects monitoring program for the Peace, Athabasca and Slave river basins. The specific objectives of this report are to: (1) Review the concepts of ecosystem health/integrity and cumulative effects assessment as well as the underlying framework and practical objectives of these approaches. (2) Critically review available literature on existing population, community and ecosystem-level approaches and associated metrics (measures) being used for assessment of aquatic ecosystem health/integrity and cumulative effects and briefly outline advantages and disadvantages of each technique. (3) Identify the types of data and information required to adequately assess and monitor aquatic ecosystem health/integrity and cumulative effects in large northern rivers. (4) Assess the applicability of these approaches and metrics to the Northern River Basins Study and, where possible, recommend approaches that could potentially be employed in the assessment and long-term monitoring of aquatic ecosystem health/integrity and cumulative effects within the Northern River Basins.

The concept of ecosystem has itself been variously defined but is most commonly thought of as a collection of interacting organisms and their physical environment. The difficulty in defining an ecosystem rests not with what it contains, but with what constitutes its borders. All ecosystems are clearly impacted by extrinsic factors. Even those who argue the globe is the only true ecosystem exclude the sun and its radiation, a clearly important input into the ecosystem. In practice, ecosystem boundaries are usually determined on the basis of the question(s) being posed and practical or political considerations. In the case of the NRBS, the ecosystem under study is comprised of those parts of the Peace, Athabasca and Slave river basins contained within the Province of Alberta and the North West Territories and is largely confined to the mainstem and major tributaries of these rivers. This restricted definition is an operational one imposed by logistical and financial limitations. It serves to provide a framework within which questions can be posed, but it recognizes the importance of other inputs and processes.

2.0 ECOSYSTEM APPROACH AND ECOSYSTEM HEALTH

2.1 ECOSYSTEM APPROACH TO ENVIRONMENTAL MANAGEMENT

The concept of ecosystem health, or ecosystem integrity, and the use of an ecosystem approach in the monitoring and management of natural systems has been advocated in various forms over the last fifty years. Only in the last two decades however, has this approach been developed and grown sufficiently popular to begin to receive serious attention from government, managers, and researchers (Haskell *et al.* 1992).

The traditional approach to setting environmental regulations, particularly in North America, has been based largely on the assessment of chemical concentrations within receiving environments. One drawback of this approach is that it has paid little attention to biological or ecological structure and function within those environments (Reynoldson and Metcalfe-Smith 1992). The recent movement toward an ecosystem approach to environmental assessment and monitoring represents a major shift in emphasis away from a chemical/physical approach toward one that recognizes: (1) the complex nature of interactions that occur at a variety of levels within the environment; (2) the fact that human populations (and their activities) constitute an important component of that environment and that they cannot be viewed as being separate and apart from it; and (3) the need for human populations to make use of natural resources in a more sustainable fashion (Marmorek *et al.* 1993). Although specific definitions of the ecosystem approach may vary, most contain three key traits: (1) an emphasis on the collection and synthesis of integrated knowledge of ecosystem structure and function; (2) a holistic perspective, interrelating systems at different levels within the ecosystem; and (3) an attempt to develop management strategies that are ecological, anticipatory and ethical (Christie *et al.* 1986; MacDonald Environmental Services Ltd., 1994).

Among the most important benefits of an ecosystem approach is the explicit recognition of human populations as being part of the ecosystem. This recognition has two major consequences for society in general, and ecosystem managers and researchers in particular. First, it serves to dispel the popular western myth that man (i.e. human populations) somehow exists outside of nature. Humans are as much a part of the natural system as is any other component and their health, economic activities and general interests should be viewed in the context of the entire ecosystem. Second, the ecosystem approach dictates that, by necessity, non-scientists, including government officials, representatives from industry, fisheries and agricultural, as well as recreational and subsistence users and the general public be directly involved in the formulation of policy regarding the management of ecosystems. The involvement of all these interest groups (stakeholders) in the development of ecosystem management goals and objectives represents a daunting and complicated task. However, such an approach is not only essential (i.e. all stakeholders have access to all information) but represents the best hope for the development of an ultimately successful ecosystem management program (Regier 1992).

The ecosystem approach also differs from the more traditional approaches to environmental management in that it has the potential to provide a framework for the long-term monitoring and protection of the ecosystem. Here again, stakeholders provide valuable input in the establishment of general ecosystem goals and the refinement of these goals into more specific ecosystem objectives. Stakeholders would also play a role in defining the "desired" state of the ecosystem and in balancing (and better measuring) the costs and benefits associated with future development and/or current remediation or restoration. Finally, the ecosystem approach also considers the state of the entire ecosystem. This is in marked contrast to more traditional approaches that examined only specific components (e.g., the area immediately downstream from a proposed pulp mill) without regard for effects at other levels or scales (both spatial and temporal).

The utility of the ecosystem approach is now widely recognized and is rapidly becoming the favoured approach for environmental management and ecosystem assessment in both North America and Europe (Marmorek 1993). It is also the approach explicitly favoured by the NRBS (NRBS 1993), as well as this report.

2.2 ECOSYSTEM HEALTH

Implicit in the concept of an ecosystem approach is the desire to maintain a managed ecosystem at some adequate level of function or health. This concept of ecosystem health, and its obvious analogy with human health, has broad intuitive appeal and has come to be widely used by managers, researchers, and members of the general public. Consequently, there now exists a considerable literature exploring the philosophical, economic and scientific implications of the concept of ecosystem health (see Costanza *et al.* 1992).

Unfortunately, the concept of health is itself, difficult to define (Calow 1992), and the development of precise definitions of ecosystem health and ecosystem integrity is particularly problematic (Haskell *et al.* 1992). Indeed, many researchers have dealt with the problem by simply avoiding all attempts at a definition. Others, finding little utility in the concept, have chosen to avoid using it all together.

Much of the confusion concerning the use of ecosystem health arises from the variety of ways in which the term health can be employed (Calow 1992). Costanza (1992) has identified several contexts in which the term ecosystem health has been used. (1) *Health as homeostasis*, or the ability of a system to maintain itself within a range of "normal variation". Within this definition any deviation beyond the normal range is deemed a decrease in health. Problems arise however, in defining what constitutes "normal variation" within an ecosystem, particularly in light of the dynamic and ever-changing nature of these systems. (2) *Health as the absence of disease*. The problem with this approach is that the definition of disease, "any departure from health", returns us to the starting point and is thus circular. (3) *Health as diversity or complexity*, assumes that more complex and diverse systems are necessarily more stable and healthy. Any reduction in diversity or complexity within an ecosystem is thus considered a reduction in health. The evidence for such a linkage is however, far from conclusive (see below). (4) *Health as stability or resilience*, refers to a systems ability to resist perturbation or to recover quickly from perturbation. Unfortunately, this definition does not speak to the system's organization or activity. Nor does it address the fact that systems are generally not static, rather, most ecosystems continually experience change. (5) *Health as vigour or scope for growth*, measures both the system's activity and resiliency. However, vigour has not been clearly defined. It should also prove difficult to measure, and will probably vary as a function of the level or scale at which it is studied. Moreover, it is not clear that a system's ability or opportunity for growth is a necessary indicator of health. (6) *Health as balance between components* has been invoked as a useful measure but there is little *a priori* reason to assume that such a balance exists. In addition, it would still be necessary to develop some other measure of health to determine when a system was out of balance.

Clearly the different uses of the term ecosystem health together with the common failure to explicitly state which use of the term is being employed has greatly confused the issue. Despite these drawbacks, or perhaps because of them, the concept has gained a firm foothold in scientific literature (as is evidenced by the recently created Journal of Aquatic Ecosystem Health) as well as in the discussions and writings of ecosystem managers and the general public. Given the concept is not likely to disappear, it would behove scientists and managers to develop a generally agreed upon working definition that will allow progress in the development and attainment of ecosystem goals and objectives.

At a recent workshop (Costanza *et al.* 1992), leading ecologists, philosophers, economists and social thinkers examined the philosophical, theoretical and applied aspects of ecosystem health as well as its general utility and implications for society and for ecosystem management. Workshop participants also recognized difficulties in defining ecosystem health but over the course of the workshop developed an operational definition as follows: "An ecological system is healthy and free from "distress syndrome" if it is stable and sustainable - that is, if it is active and maintains its organization and autonomy over time and is resilient to stress." (Haskell *et al.* 1992, p. 9). This definition is applicable to all complex systems and identifies four key traits that must be possessed by a healthy ecosystem: (1) sustainability, (2) activity, (3) organization and, (4) resilience. The definition is not intended to serve as a final and complete definition of ecosystem health, rather its purpose is to more explicitly state our current understanding of the concept and to serve as a starting point for future research and discussion in this emerging and multi-disciplinary field.

This operational definition has the advantage of formalizing and incorporating several key concepts relating to ecosystem health and also provides an excellent starting point for the development of a long-term strategy for the study and improvement of ecosystem health/integrity. Unfortunately, while it recognizes the importance of ecosystem characteristics such as sustainability, resilience and autonomy, these characteristics may, themselves, be difficult to define in a practical context. The operational definition also provides no direct information as to the best way to measure these characteristics or to incorporate them in a meaningful way into specific ecosystem objectives. It should also be noted that the desired levels of these characteristics will be influenced by the temporal and spatial scale at which they are measured as well as by the perceived priorities of stakeholders (see below). The ability of any system to maintain its own activity, organization and resilience will ultimately dependent on a suite of factors and processes unique to that system, Therfore, while the operational definition developed at the workshop provides a first step on a multi-disciplinary research program it is unlikely to be of direct, immediate use to those charged with the task of managing ecosystems.

Rather than concentrate solely on specific definitions, Calow (1994) has identified three general theoretical frameworks within which researchers have attempted to develop the concept of ecosystem health. In the first case, ecosystems are viewed from a holistic point of view. In other words, ecosystems are thought of as essentially super-organisms maintaining some optimum (or homeostatic) state or alternatively, as systems that maintain a steady equilibrium stable state. Among the general public, this view is best represented by the "balance of nature" arguments. In the second case, healthy ecosystems are defined in an anthropocentric sense in which the health of an ecosystem is determined by its ability to provide the services demanded of it by the stakeholders. Under this scenario an ecosystem capable of satisfying the economic and aesthetic demands of a human society is deemed healthy. In the third case, ecosystem health is defined in a pragmatic sense. This approach does not seek to develop a general definition of ecosystem health but rather, combines the best available scientific knowledge with subjective expectations of the ecosystem as determined by stakeholders. The objective is to develop a pragmatic, operational view of the desired structure and function of the ecosystem. Under this approach, what constitutes a healthy ecosystem may change from place to place as the stakeholder input and available scientific information changes.

2.2.1 Organism/Stable State View

Calow (1992) has pointed out that the organismic view of ecosystem health implies some sort of goal-directed, or teleological, behaviour as well as the existence of feedback loops to mediate this behaviour. However, this view of ecosystems as organisms is flawed. Ecosystems are not organismic in the sense that they can be conceived of as a single unit. They do not reproduce, compete amongst themselves for limited resources or possess a genotype on which natural selection can act. Given that ecosystems are not subject to natural selection there is no mechanism by which they could evolve (in the traditional sense) toward an optimum state. Furthermore, there would exist no evolutionary advantage to maintaining an optimum state and indeed, there is no good evidence that an optimum ecosystem state even exists. Various components and sub-components of ecosystems may and will be subject to natural selection and will possess optimum states, but while ecosystems themselves may change over time they possess no optimum "healthy" state (Calow 1992, 1993).

The "equilibrium/stable state" view suggests that the various components and subcomponents of the ecosystem interact in such a way as to develop an equilibrium state(s) that is stable over time. The existence of these stable states can be used to designate an ecosystem as being healthy. Calow (1994) points out three difficulties with this approach. First, the ability of an ecosystem to maintain a steady state will be dependent on the variability of the environment containing that ecosystem. In other words, stable states are context dependent. This being the case, a measure of how stable an ecosystem (or components within that ecosystem) is cannot be used to reliably determine the state of ecosystem health. Second, there is considerable theoretical and empirical evidence for intrinsic and extrinsic processes that would prevent ecosystems from maintaining long-term stable states. The phenomena of succession and of predator-prey dynamics are perhaps two of the best known examples of processes which drive communities or ecosystems away from a constant steady state. Third, the equilibrium/stable state view fails to recognize the importance of stochastic or random events that disrupt community or ecosystem structure and allow for reinvasion. Indeed, disturbance, particularly at intermediate levels, is widely recognized as an important determinant of ecosystem structure and function (Connell 1978; Sousa 1979; Resh *et al.* 1988).

The remaining two general theoretical frameworks discussed by Calow are similar in that they both rely on a human-based definition of what constitutes a healthy ecosystem. Both of these latter approaches to the concept of ecosystem health also have important implications for issues of ecosystem medicine as discussed by Lee *et al.* (1982); Rapport *et al.* (1985) and Rapport (1992a,b,c). The issue of ecosystem medicine and clinical ecology will be discussed in greater detail below.

2.2.2 Anthropocentric View

The anthropocentric view argues that stakeholders should identify those services they require from the ecosystem and subsequently devise management goals and objectives that best fulfil those requirements. A healthy ecosystem, as defined by this approach, is one that is capable of providing the identified services at some acceptable and agreed upon level. An anthropocentric definition of ecosystem health explicitly recognizes that the ecosystem is to be managed according to the perceived needs, goals and priorities of the relevant human population. Drawbacks of an exclusive reliance on this approach include: (1) Given the contrasting requirements of different stakeholders it may not be possible to simultaneously maximize the ecosystem's ability to supply all desired services. Under these circumstances, it will be necessary for all stakeholders involved to develop a series of compromises as to the nature and extent of services they desire from the ecosystem. (2) Even if all stakeholders agree on a suite of services required of the ecosystem it may not be possible to manage the ecosystem so as to maintain such a state. In this case an ecosystem would be deemed unhealthy solely because of its inability to supply all stakeholders with all the services they desire. (3) Managing an entire ecosystem solely from the perspective of human goals raises clear ethical concerns (Calow 1994) and is a direct contradiction of the ecosystem approach now favoured by most levels of governments in North America and in Europe.

2.2.3 Pragmatic View

The final approach discussed by Calow combines the best available empirical and theoretical scientific knowledge of the ecosystem with subjectively derived (through stakeholder input) expectations of what the ecosystem should look like. The objective of this exercise is to develop a pragmatic definition of ecosystem health. It is this approach that is most often employed by ecologists when assessing ecosystem health or integrity, and which most closely resembles the diagnostic approach used in human medicine. It should also be noted, that because this approach (like the anthropogenic approach) relies on stakeholder input and scientific information, the expectations of what the ecosystem should look like may change as societal priorities and the state of scientific knowledge change. As a consequence, the basis on which an ecosystem is judged to be either healthy or unhealthy may change from region to region and may change within a region over time.

As with other approaches discussed above, the challenge in pragmatically defining ecosystem health is to identify ecosystem norms and to reliably detect unacceptable movement away from those norms. Unfortunately, environmental norms for a given site, thought to be exposed to anthropogenic stresses, have often been determined by comparison of the supposed impacted site to a similar unimpacted, or reference site. Comparison of sites of interest to reference or control sites is in fact, the major diagnostic tool currently used in biomonitoring programs (see following sections). This approach is typically constrained by the environmental similarity of the two sites independent of the impact (i.e. the quality of the control or reference site) and by the fact that while such comparisons can assess differences between or among sites, they cannot address the question of what is, or was, responsible for the observed differences (i.e., they cannot determine causality). However, recent and ongoing improvements in study design and sampling methodology (e.g. QA/QC procedures), an inclusion of controlled and rigorously designed experiments in the evaluation of ecosystem health, and the development of more powerful statistical approaches to deal with environmental data have greatly improved our understanding of ecosystem structure and function and of the underlying mechanisms responsible for maintaining that structure and function.

It would appear therefore, that the pragmatic approach to ecosystem health provides the greatest utility to those tasked with managing ecosystems. This approach does not attempt to develop a precise and general definition of ecosystem health, and thus avoids the very real problems and challenges faced by those that do (Costanza *et al.* 1992). Although it develops a somewhat arbitrary and region-specific definition of ecosystem health, this approach not only makes full use of the best scientific information available but is also capable of making subjective assessments of ecosystem health based on this information. Importantly, this approach is sufficiently flexible to allow for the incorporation of new information, changes in societal priorities, improvements in monitoring techniques and/or refinements in theoretical understanding as they become available. Defining ecosystem health within a pragmatic framework also facilitates the development of ecosystem goals and objectives, and permits the direct involvement of stakeholders in the development these goals and objectives and in their further refinement into specific management objectives.

One potential criticism of the pragmatic approach to the assessment of ecosystem health is its failure to explicitly consider such key issues as sustainability, activity, organization and resilience (Costanza *et al.* 1992). As discussed above, these issues are central to an adequate understanding of ecosystem health and are the subject of considerable research. Unfortunately, the pressing need to develop effective ecosystem management tools and protocols means that management strategies that can operate at the level of the ecosystem must be developed while research continues. While major theoretical investigations of ecosystem health are clearly beyond the scope of the NRBS, an awareness and careful consideration of research in this area is strongly recommended. This is particularly important because issues relating to sustainability, activity, organization and resilience are, and will continue to be, important considerations in determining the specific ecosystem goals and objectives within which the pragmatic approach to ecosystem health monitoring and assessment would operate.

2.3 CLINICAL ECOLOGY

The parallels between ecosystem health and human health have been alluded to above and may ultimately provide a useful framework within which issues of ecosystem health and integrity could be examined. David Rapport (1992 a,b,c) has been particularly active in pursuing this analogy and has even gone so far as to suggest the development of an entire research field known as ecosystem medicine or clinical ecology.

The field of clinical ecology seeks to further develop and extend the analogy between research into ecosystem and research relating to human health issues. As with the field of human medicine, research into ecosystem medicine would have a number of primary objectives including: (1) the development of criteria that could be used to measure and/or define acceptable levels of function (i.e., an operational definition of ecosystem health/integrity); (2) the development of a suite of diagnostic tools capable of assessing general ecosystem condition or health; (3) the development of a suite of diagnostic tools capable of detecting, in a timely and informative fashion, departures away from a "healthy" or acceptable state; and (4) the development of a variety of tools and techniques to ensure ongoing adequate "health" or, in the case of a demonstrated negative anthropogenic stress or impact, a return to a healthy state once the problem had been adequately diagnosed.

Ecosystem medicine is based on the premise that much of the scientific methodology developed in the field of human medicine can be applied directly, or indirectly, to issues of ecosystem health. There are however, important differences between the two fields. Ecosystem medicine differs from human medicine in that it does not take the view that its objective is to treat single individuals (i.e., an ecosystem) so as to maintain some pre-defined optimal state. In other words, ecosystem medicine, or clinical ecology, does not advocate a super-organismic view of the type criticized by Calow (1992). Ecosystem medicine also differs from human medicine in that it does not place primary emphasis on the cure of the "disease" or the perceived movement away from health. Rather, it stresses an adequate understanding of ecosystem structure and function as well as a knowledge of the boundaries within which ecosystems may be said to act normally. In short, ecosystem medicine does not attempt to treat ecosystems as if they were simply diseased patients. Rather, it makes use of an understanding of human health issues developed over thousands of years to inform the process by which managers and researchers assess ecosystem health, diagnose deviations from a desired state, and even determine the ideal characteristics of a "healthy" ecosystem.

Within the field of ecosystem medicine the definition of health shares much in common with Calow's (1994) pragmatic approach. Rapport argues that ecosystem health, as is the case with human health, is largely determined by an integration of the best available scientific information and some subjective evaluation of what the system "should look like". Implicit in this definition is the role of societal values and stakeholder agendas in determining what constitutes a healthy ecosystem.

If ecosystem health is to be judged on the basis of both objective and subjective criteria then it is important to measure health in a variety of different ways and a variety of different levels, since there is unlikely to be a single optimum "healthy" state (Rapport 1992a). It is also important to note that because the definition of health and the way in which it may be measured is subject to change those attempting to monitor and preserve ecosystem health must develop flexible programs capable of operating at different levels and responding to changes in available techniques and societal values.

Diagnosing ecosystem health is further complicated by the tremendous natural variation (both temporal and spatial) observed in many ecosystems. This high degree of natural variation together with limitations on our understanding of basic ecosystem theory, as well as structure and function, makes it difficult to identify the norms within which a given ecosystem operates. The challenge of ecosystem management is to develop measures sufficiently sensitive to provide early warning of departures away from a desired state, but not to confuse acceptable variance in structure and function with a process requiring remedial action. In the terminology of ecosystem medicine, natural variation may serve to mask "symptoms" important to the detection of "pathologies" within the system and may also give rise to false negatives that could lead managers to intervene in ecosystem function when it was unnecessary or even harmful to the system. Finally, care must be taken in choosing what to measure within the ecosystem. For instance, many ecosystem level processes fail to show evidence of "pathology" until it is well advanced (Rapport 1992a) and thus will serve as poor "diagnostic" tools (see following sections for examples). Appropriate diagnostic indicators should be holistic, readily measured at a variety of spatial and temporal scales, provide early warning of a "pathology" within the system, and be diagnostic of specific causes.

Interestingly, Rapport also argues for the development of indicators that focus on health rather than evidence of pathologies (Rapport 1992b,c). Because it will often be easier to identify and quantify undesirable states relative to desirable ones, these indicators may be particularly difficult to develop but their utility would be very great. Indicators that focus on the health of the ecosystem would serve to identify critical feedback mechanisms necessary to maintain ecosystem integrity and measure the effectiveness of those mechanisms. By employing indicators of this type clinical ecology could eventually develop a suite of tools allowing it to practice "preventative" as well as "curative" ecosystem medicine.

While the area of clinical ecology or ecosystem medicine may provide a theoretical framework within which issues of ecosystem health and integrity could be examined the field has thus far failed to provide the specific tools required to satisfy its objectives. There is no question that feedback mechanisms and multi-scale holistic health indicators of the type described above would be very useful in assessing and monitoring ecosystem health. Unfortunately, empirical investigations seeking to identify, properly measure, and validate these indicators lags behind the theoretical research and discussion.

To further complicate the issue, there is also a question of whether ecosystem level measures of the type advocated by Rapport are sufficiently sensitive to provide useful tools with which to monitor ecosystem health (Schindler 1987, 1988, 1990; see below). There may also be considerable difficulty in extrapolating from observations obtained at any one spatial or temporal scale to other scales, an effect that may limit the diagnostic utility of any single ecosystem health indicator.

The role of indicators in the development of biomonitoring programs designed to assess ecosystem health and cumulative effects impacts will be discussed in greater detail in a subsequent section of this report.

2.4 CONCLUSIONS

The preceding discussion has hopefully illustrated that while the concept of ecosystem health is difficult to define it is integral to any application of the ecosystem approach to environmental management. Concepts such as ecosystem approach and ecosystem health also provides a useful theoretical framework, within which ecosystem management goals and objectives can be identified and developed.

Although there are several ongoing attempts to develop a precise and unambiguous definition of ecosystem health the concept remains elusive and difficult to define. Of greater utility is an approach that determines ecosystem health on the basis of an objective evaluation of the best available scientific information in concert with a more subjective evaluation of health based on available information and societal values. This approach retains flexibility while facilitating the development of specific ecosystem goals and objectives required by managers. Management of ecosystems may be further assisted by an understanding and appreciation of approaches taken to human health issues.

In the context of the NRBS, this pragmatic approach to the evaluation of ecosystem health would involve several steps. First, a synthesis of historical and current studies would allow for the development of a single, comprehensive data base that would fill information gaps and result in an improved understanding of the basic structure and function of these systems. Second, this data base should be related directly to areas of management concern (human health, ecological condition, economic concerns, etc.) to identify components within the ecosystem that may be particularly at risk from anthropogenic activities. This is the approach currently being developed within the Synthesis/Modelling Component of the NRBS. It involves the construction of an information matrix that identifies and categorizes various potential indicators and relates these indicators to general concerns identified by the stakeholders in the NRBS guiding questions and through the course of public consultation. In the final step, the selection of specific indicators to be incorporated into the cumulative effects assessment program involves a process by which the concerns and goals identified by stakeholders are further refined into specific management objectives. Potential indicators are then evaluated in terms of their ability to fulfil those objectives in an efficient and cost-effective manner.

3.0 BIOMONITORING

3.1 INTRODUCTION

As the extent and complexity of anthropogenic impact on the environment increases so does the need to develop effective management criteria that can be used to maintain current levels of ecosystem structure and function and, where necessary and possible, take remedial action in those systems deemed to have been unacceptably impacted. Essential to the development of any effective ecosystem management strategy is the development and implementation of a suite of appropriate biomonitoring techniques. A properly designed biomonitoring program would be based on, and contribute to, the existing data base describing the general nature (i.e., structure and function) of the ecosystem being managed. Such a program would also provide early warning of changes to that system, and ultimately provide information as to the causes of those changes and the steps required to restore the ecosystem to some acceptable level of structure and/or function (ecosystem health).

The last several decades have witnessed a tremendous increase in the number and types of biomonitoring techniques available to ecosystem scientists and managers. Current research in aquatic biomonitoring is being conducted by researchers in academic institutions as well by those in various levels of government and industry. Specific research projects span fields as diverse as genetics, paleolimnology, biochemistry, physiology, toxicology, taxonomy, multivariate statistics, and genetics as well as in basic ecology and systematics (Burton 1992; Johnson *et al.* 1993; Rosenberg and Resh 1993). Clearly, a complete and detailed review of techniques being used for the assessment of aquatic ecosystem health is beyond the scope of this report. Rather, I propose to provide a basic overview of the most commonly employed aquatic biomonitoring techniques and to assess their usefulness to the NRBS.

3.2 OBJECTIVES

Specific objectives of this section of the report are: (1) to provide a discussion of some of the general issues involved in the development of any ecosystem-level biomonitoring program, with particular reference to issues facing the NRBS; (2) to provide a brief description of the major biomonitoring techniques currently in common use and to discuss the advantages and shortcomings of each and; (3) to provide suggestions as to the best approach for developing a biomonitoring framework for the NRBS.

Owing to the nature of the field and my own background many of the examples in the following sections will be drawn from studies on benthic macroinvertebrates. This should not be interpreted as a suggestion that biomonitoring programs employing macroinvertebrates are necessarily superior, or that biomonitoring techniques developed for benthic macroinvertebrates cannot be modified for other taxa. Indeed, in most cases the techniques described are not dependent on any one taxa and reference to specific studies employing benthic macroinvertebrtaes should be viewed as examples only. In the final analysis, the most appropriate taxonomic group (or groups) to monitor will be a function of the specific ecosystem goals and objectives of the biomonitoring program.

3.3 GENERAL ISSUES

3.3.1 Introduction

The specific design of any biomonitoring program will be contingent upon the ecosystem goals and objectives of the program itself, as well as on inevitable financial and logistic limitations. There are, nevertheless, basic issues which should be considered in the development of any biomonitoring program designed to assess ecosystem health and cumulative impacts. What follows is an identification and brief discussion of several of these general issues. An awareness and understanding of these issues will aid greatly in the design of an effective biomonitoring program and will assist in the identification of knowledge gaps in our current understanding of the ecosystem.

3.3.2 Basic Ecology

An explicit objective of the NRBS is to acquire a baseline data set pertaining to the basic ecology of the Peace, Athabasca, and Slave river basins. Such information is essential, not only because it provides an understanding of the ecosystem structure and function as it currently exists, but also because it will provide a reference point for future comparisons. As pointed out by Johnson *et al.* (1993) it is impossible to apply knowledge that one does not have and the success of any biomonitoring program or ecosystem management strategy will be largely constrained by the understanding of the basic ecology of the system under study.

Unfortunately, there are considerable gaps in our current knowledge of the ecology of the Peace, Athabasca, and Slave river basins. These knowledge gaps are reflective of the difficulties associated with working in these systems and of a more general lack of information on the ecology of large rivers, particularly large northern rivers. In the case of the NRBS much of the baseline data required to develop a biomonitoring program are currently being collected and/or synthesised. The results of these activities will not be made available in a complete form until the study is nearly concluded. Knowledge gaps relating to the basic ecology of these systems greatly complicates the development of a biomonitoring framework required to assess ecosystem health and cumulative effects within these basins. It should however, be possible to provide a general approach to the development of such a framework even if the specific biomonitoring techniques and procedures cannot be incorporated into the framework until data currently being collected are made available.

The importance of understanding the structure and function of the ecosystem cannot be over emphasised. It is this understanding which determines our view of the system and provides a context within which all management priorities and objectives are developed. Gaps in this understanding could result in a failure to identify key issues or in the misdirection of time and effort.

At a more pragmatic level, an adequate understanding of ecosystem structure and function is essential in order to (1) accurately trace the path and fate of contaminants once they are introduced to the system, (2) identify those components of the ecosystem most likely to be affected by such an introduction, and (3) predict the overall effect of contaminants or groups of contaminants on the nature of individuals, populations, communities and the ecosystem. An understanding of the basic ecology of the system will also be important in predicting the long term consequences of any observed change in community structure or function; in determining the underlying mechanisms responsible for the observed changes; and in identifying those species, or groups of species, that play an important ecological role in the maintenance of the community and/or ecosystem as a whole.

Information relating to the basic ecology of the ecosystem under study may also be a prerequisite for the successful application of commonly used biomonitoring techniques. Several community-based monitoring techniques described in the following sections require that all individuals collected be identified to the level of family genus, and in several cases species. Such an approach presupposes a detailed taxonomic knowledge of the species being collected and can (particularly in the case of benthic macroinvertebrates) entail considerable costs. Similarly, knowledge of the ecology of individual species is required before they can be properly assigned to different functional groups or used in saprobic or biotic integrity indices. If individual species are to be employed as bioindicators within the biomonitoring program their taxonomy and distribution must be well understood, as must their basic ecology and response to perturbation (IJC 1991; Cairns *et al.* 1992; Johnson *et al.* 1993). Knowledge of the movements, ecology, and population structure of potential bioindicators is also important in the development of techniques relating to the locating and sampling of such species. Finally, the utility of chronic toxicity tests and bioassays that attempt to predict environmental effects is dependent on the selection of ecologically relevant endpoints (survival, growth, fecundity, performance), and a knowledge of the ecological roles and trophic interactions of the test species (Burton *et al.* 1992; La Point and Fairchild 1992; Buikema and Voshell 1993).

Ideally, environmental monitoring programs would have at their disposal, accurate and complete information as to the basic ecology of the ecosystems being monitored. Such a data base could be used to clearly identify the most appropriate bioindicators, and to identify those ecosystem components most sensitive to perturbation as well as the manner in which different components within the system interact. Unfortunately, limitations on human and financial resources as well as on our ability to understand complex ecological processes preclude this possibility. In reality, programs such as the NRBS face the challenge of having to synthesis available knowledge and fill large information gaps in baseline data, while at the same time determining the impact of anthropogenic activities on the system and developing a framework for ongoing ecosystem health and cumulative effects monitoring.

Despite the limitations described above, a carefully designed and executed biomonitoring program will nevertheless be capable of generating a data base providing information on the basic structure, function and ecology of the system under study. The existence of such a data base has several important advantages: (1) An understanding of the basic ecology provides the context within which ecosystem goals and objectives are formulated. (2) An adequate and accessible data base would provide researchers and managers with the flexibility required to apply different interpretative techniques to the same data set and to select the one that best meets their objectives or that proves to be most cost effective. (3) Improvements in biomonitoring techniques could be retroactively applied to "quality" data already collected and synthesized. (4) The existence of a long-term, carefully constructed data base will facilitate the detection of important ecological trends, may provide early warnings of changes to the ecosystem and will provide a background against which the progress of remediation efforts can be judged.

3.3.3 Study Design

One of the major goals of any biomonitoring program is to use the patterns of distribution and abundance of organisms observed within the ecosystem to determine the state of that ecosystem and to detect change within that system. The extent to which this goal is met is thus dependent on the ability to (1) identify those components of the ecosystem that require measurement; (2) properly measure and describe those components; (3) compare and contrast those measures at a variety of spatial and temporal scales; and (4) relate these observed patterns to corresponding patterns in physicochemical variables. The development of appropriate study designs is critical to this process and will facilitate management objectives by assuring the proper collection of relevant data, the elimination of confounding effects and the selection of appropriate analyses (Norris and Georges 1993).

Spatial and temporal variation in the distribution and abundance of organisms is often considerable, even in the absence of any disturbance. It is therefore important that environmental variability and its effects on sampling accuracy and precision be accounted for both in study design and in data analyses. The past two decades have witnessed considerable progress in several areas of study design and data collection.

Although more progress is required, there have been considerable improvements in, and greater standardization of, field sampling and collection techniques (Downing 1979; Cuffnet *et al.* 1993a,b; Gibbons *et al.* 1993; Meador *et al.* 1993a,b; Porter *et al.* 1993; Resh and McElravy 1993). It is also now generally recognized that even small habitat differences among sites can be a major source of natural variation. Sampling protocols should thus include habitat characterization and measurements of all important and relevant physicochemical variables (Norris and Georges 1993).

Improvements in study design such as the development of the BACI (Before After Control Impact) approach (Green 1979), a recognition of the importance of sample replication and statistical power, and the development of Quality Assurance/Quality Control protocols have helped to ensure that the appropriate types of accurate and precise data are collected and properly handled. The increased use of powerful univariate and multivariate statistical techniques will permit researchers and managers to better identify pattern within the data set and to better discriminate between natural and stress induced variation in the abundance and distribution of organisms (Green 1979; Dixon and Newman 1991; Jackson 1993; McBride *et al.* 1993; Norris and Reynoldson 1993). These techniques are also useful in the generation and rigorous testing of hypotheses relating to the underlying causal mechanisms responsible for the observed variation (Norris and Georges 1993).

3.3.4 Scale

Issues of scale in the design of biomonitoring programs are closely related to those of appropriate sampling design. Scale is an important consideration, not only, as discussed above, from the perspective of adequately describing and sampling a system as large the Peace, Athabasca and Slave river basins, but also from the perspective of interpreting and identifying spatial and temporal pattern in the data once they are collected. Indeed, a growing number of researchers have argued that the problem of pattern and scale is rapidly emerging as one of the central problems in population ecology and ecosystem science (Fox 1992; O'Neill *et al.* 1992). These researchers also argue that issues of scale and pattern represent an important bridge between theoretical and applied ecology (Levin 1992), and should play an important role in the ongoing monitoring and assessment of ecosystem health and in the development of biomonitoring programs.

In the first instance, developing a large-scale monitoring program such as that required by the NRBS presents considerable logistic as well as theoretical difficulties (O'Neill *et al.* 1992). Questions that arise include (1) What are the extent and intensity of sampling efforts at different spatial and temporal scales required to adequately describe these system? (2) Are there specific components of the ecosystem that are particularly vulnerable to anthropogenic stresses, or that are key to ecosystem function? (3) Can managers extrapolate from patterns observed in one area, time, or level of organization within the ecosystem to other areas, times, or levels of organization within the same system?

In many cases the question of what scale is most relevant can be addressed on the basis of available background data on the physical, chemical, and biological nature of the ecosystem and of the nature, source, and timing of stresses that act on the system. If such information is not available, then biomonitoring programs should be designed in such a way as to begin to construct such a data base and should contain the flexibility required to respond to new information as it becomes available.

The problem of deciding the most appropriate scale (spatial, temporal and organizational) at which to observe pattern is further complicated by the effect of scale on the interpretation of pattern once observed. Because each species or group of species experiences the environment at a unique range of scales, the scale of observation chosen by the researcher or manager will influence the description of pattern. It is thus necessary to ensure that researchers are careful to to chose a scale of observation appropriate to the question being asked (Levin 1992). In other words, specific patterns observed within the environment are largely a function of the scale at which workers choose to make observations and a change in the choice of scale may well change the pattern observed.

This observation has important consequences for the design of biomonitoring programs. Measurements collected at the level of the individual (or in single species toxicity tests) may be appropriate for examining the short-term behaviours of individuals but may not be appropriate for examining populations, communities or whole ecosystems (Buikema and Voshell 1993). Similarly, patterns observed within communities may contain little information on the response of individual species or of the entire system (Cooper and Barmuta 1993). Finally, in long-term studies of whole lake ecosystems, Schindler (1987, 1988, 1990) has demonstrated that significant changes in species composition and community structure may not be reflected by changes in ecosystem level processes. This suggests that monitoring at the level of the ecosystem itself may not provide the data required to properly assess ecosystem condition or to detect changes in ecosystem structure and function.

Issues of scale and pattern will continue to complicate the interpretation of biomonitoring data and are clearly deserving of further investigation. Problems arising from the misinterpretation of biomonitoring data can be minimized if issues relating to scale are explicitly recognized both in the design of studies and in the interpretation of results. Confusion resulting from scale-related problems may also be minimized by: (1) giving careful consideration to the scale or scales of relevance for a particular question, (2) collecting observations from a variety of different spatial, temporal, and organizational scales, (3) being sensitive to the potential difficulties in extrapolating between scales (Cooper and Barmuta 1993), and (4) being aware of the fact that the causal mechanisms producing the observed pattern often occur at a scale below that at which the pattern is observed (Levin 1992).

In addition to these general issues relating to scale there are particular concerns of direct relevance to the NRBS including: (1) The need to develop biomonitoring tools for use at point source discharges as well as at a basin-wide level. (2) The need to account for the tremendous variability in a system as large and diverse as the Peace, Athabasca and Slave river basins. (3) the need to identify and reconcile the different scales impacted by a single point or non-point source discharge.

As an aside, Frost *et al.* (1992) have suggested that the most appropriate scale of taxonomic organization at which to select ecological indicators is one which shows minimal natural variation but is maximally sensitive to the stress of concern. In their studies of zooplankton they found that this condition was best satisfied at intermediate levels of taxonomic organization. These findings suggest that studies at the scale of guild or functional group may be represent an important level for the development of ecological indicators. As has been discussed, however, the final choice of scale will be a function of the question being asked.

3.3.5 Experimentation

A well designed monitoring program is capable of detecting pattern within the environment, identifying trends in the state or condition of the ecosystem, and, in some cases, can provide inferences as to the cause or causes of the observed trends. However, in the absence of controlled experiments properly and rigorously designed to test specific hypotheses, monitoring programs cannot determine the underlying causes of observed patterns (Clements 1991; Rose and Smith 1992).

In the past, the limitations of monitoring alone have not always been fully appreciated. For example, differences in measurements from locations obtained immediately above and below a point source discharge, such as a pulp mill, may be properly taken and analyzed but would only serve to demonstrate differences and could provide no explanation as to the cause of those differences. Differences of this type have traditionally been misinterpreted as evidence of a causal link between the presence of a point source discharge and some presumed downstream effect. In reality, additional information relating changes in measures taken to differences in the relevant environmental variables and the use of properly designed and rigorous experiments would be necessary to demonstrate any causal link between the presence of the point source discharge and the observed downstream changes.

Properly designed and executed field and laboratory experiments should play an integral role in the development and operation of biomonitoring programs. Employing a well designed experimental approach will allow managers to: (1) investigate, under replicated and controlled conditions, many important aspects of field conditions; (2) better interpret observed responses of individuals, populations and communities to environmental change; (3) calibrate and validate existing or proposed monitoring programs; (4) identify sensitive components within the ecosystem that could serve as indicators of health; (5) predict responses to possible perturbations acting on the ecosystem; (6) disentangle the direct and indirect effects of such perturbations; and (7) determine the direct and interactive effects of a variety of variables on ecological systems (Cooper and Barmuta 1993).

As was discussed in the previous section, extrapolation from experimental results to phenomena observed at other scales is often complicated and is deserving of further research. However, performing rigorous, controlled experiments designed to test specific and relevant hypotheses at a variety of different spatial and temporal scales will increase understanding of the interaction between scale and pattern. Clearly, all experiments involve some sacrifice of reality and accuracy in favour of an increase in precision, but they also provide the best opportunity to rigorously test hypotheses generated from an examination of monitoring data and to identify the causal mechanisms responsible for environmental change.

3.3..6 Multiple Impacts and Cumulative Effects

Traditional aquatic biomonitoring programs were largely developed to examine the potential effects of organic pollutants on the environment (Metcalfe-Smith 1994). However, aquatic organisms in nature are routinely exposed to a great variety of different stresses, both organic and inorganic, simultaneously. Common stressors include organic pollution (sewage), heavy metals, dioxons, furans, and organochlorines to name a few (Costan *et al.* 1993). In some cases, as with pulp mill effluent, some of the most important contaminants are thought to be as yet unidentified.

Current biomonitoring programs must therefore be sensitive to a variety of perturbation types as well as to the additive and synergistic effects of exposure to several different types of stress simultaneously.

In a similar sense, single-species acute toxicity tests in which the test organism is exposed for a brief period to a single toxicant remains the most frequently used toxicity test in aquatic biomonoitoring (Buikema and Voshell 1993). These tests clearly provide important information, particularly with regard to the development of regulations governing the release of chemicals into the environment and they will continue to play a vital role in environmental science. However, there is also a need to modify these tests so as to evaluate the effects of complex mixtures of chemicals and to examine the cumulative effects of both short- and long-term exposure to multiple stressors on individuals, populations, communities and ecosystems.

3.4 SPECIES LEVEL BIOMONITORING

3.4.1 Bioindicators

Biomonitoring at the level of individuals or species makes use of the concept of indicator species. Indicator species, as used in this report, follows the definition given by Johnson *et al.* (1993, p. 40), a bioindicator species is "a species (or species assemblage) that has particular requirements with regard to a known set of physical or chemical variables such that change in the presence/absence, numbers, morphology, physiology, or behaviour of that species indicate that the given physical or chemical variables are outside its preferred limits." These authors further argue that the ideal bioindicator would have the following characteristics: (1) Taxonomic soundness and easy recognition by the nonspecialist. This will simplify long-term monitoring, reduce the cost of such monitoring and will facilitate comparisons among sites. (2) A cosmopolitan, or at least broad, distribution. Such a distribution will facilitate comparisons among sites within an ecosystem as well as between ecosystems. (3) Numerical abundance of the indicator species may simplify collection and provide sample sizes required for quantitative analyses. (4) Low genetic and environmental variability in the indicator species. This trait would reduce the background "noise" and simplify the process of identifying the cause of variation (i.e., the anthropogenic impact. (5) Appropriate body size would simplify both sampling and sorting. (6) Limited mobility and relatively long life history. A species with these characteristics is more likely to be representative of the area in which it was collected and has integrated information over time. (7) A well understood ecology will facilitate collection, measurement and interpretation. (8) An indicator species amenable to experimental investigation will allow for an exploration of causal mechanisms.

Although thousands of indicators have been used in environmental studies (Peakall 1992), the International Joint Commission (IJC) (1991) suggests that difficulties in selecting an appropriate indicator or indicators may be reduced by recognizing the different ways indicators are used in biomonitoring programs. Individual species (or species assemblages) may be used as indicators of: (1) current ecosystem condition, (2) long-term trends within the ecosystem, (3) early warnings of change or stress acting on the ecosystem, (4) indicators that are diagnostic of particular types of stress or perturbation and (5) indicators that serve to demonstrate linkages among different components of the ecosystem.

Indicators that are to be used to assess current ecosystem condition should play an important role in maintaining the community. In other words, they should possess biological or ecological relevance. Indicators possessing ecological relevance should measure or be directly related to endpoints such as survival, growth, reproduction, or performance of some component of the ecosystem. Ideally, indicators used to assess current ecosystem condition should also be of value (or predictive of measures that are of value) to stakeholders. In other words they should posses social relevance.

Indicators used to assess current ecosystem health should also respond to a wide variety of stressors or combinations of stressors, and be readily interpretable so as to give a clear indication of required management action. The existence of an historical data base for these species will provide information on original condition of the ecosystem and aid in the setting of "target" conditions.

Sentinel animals or plants (National Research Council 1991) are those which concentrate pollutants from their surroundings, or from their food, within their own tissues. Tissue analyses of these species will therefore provide a measure of the bioavailable concentrations of pollutants within a particular area. The ability to provide information on bioavailable concentrations of pollutants suggests that sentinel animals or plant indicators would be particularly useful in assessing current ecosystem condition. The challenge, as always, is to identify those species within the ecosystem that could best serve as logistically feasible and cost-effective sentinel animals or plants and whose ecology and physiology is sufficiently well understood to allow them to play such a role.

Indicator species used to assess long-term trends within the ecosystem must possess those characteristics found in indicators of current ecosystem condition but should also possess additional characteristics. Indicators of long-term trends should possess what the IJC (1991) refer to as continuity, that is, the ability to measure the same response variable at the same level over a long period in time.

Changes in the ability to detect the same response over long periods of time (eg. historical changes in the detection limits of several types of toxic substances within the Great Lakes) will compromise the ability of managers to detect real change in the environment. The existence of an historical data base will also be particularly useful for any species that is used to assess long term trends.

Species that are to be used as early warning indicators are very likely to possess characteristics quite different from those used to assess ecosystem condition. Early warning indicators need not possess biological and social relevance or continuity. In other words, early warning indicators need not give indication of unacceptable change at the level of the community or ecosystem. Rather, the ideal early warning indicator would be sensitive to perturbation and, although readily measured, would not necessarily be numerically abundant (i.e., would not be a numerically important component of the community being examined), would display a rapid response to stressors and would be of less immediate value to stakeholders.

The strength of early warning indicators lies in their potential to provide information to researchers and managers in a timely manner. In many cases impacts at the level of the community (eg. changes in age or size-class structure within fish communities) or ecosystem (changes in total production) may not be apparent until considerable damage has occurred. As a consequence of these time lags, managers without access to early warning indicators, maybe forced to undertake involved and expensive remediation actions that could have been avoided had an early warning indicator been in place.

Diagnostic indicators would be used to identify specific causative agents following the identification of a problem. Diagnostic indicators should thus be very specific in their response to stress, and managers would, in all probability, require a suite of stress-specific diagnostic indicators. Biochemical and physiological processes show particular promise for use as diagnostic indicators (see below). Although a biomonitoring program may include a large number of diagnostic indicators most would only be employed after a problem has been identified. This more restricted use of indicators suggests that even some of the more expensive and difficult to employ techniques (eg. various biochemical/physiological indicators) could be cost effective when used to investigate specific problems.

An appropriate diagnostic indicator could be very site-specific and possess a restricted distribution under the scenario where the potential impact of specific point source discharges was of primary concern. Alternatively, diagnostic indicators might integrate information at the level of basin or ecosystem. Sentinel species which bio-accumulate specific pollutants could also serve as diagnostic indicators. Only direct tests or experimentation are capable of identifying the cause of perturbation, but diagnostic indicators can be highly suggestive. As discussed above, the appropriateness of any given indicator (diagnostic or otherwise) will also be a function of the scale of relevance for question being posed.

Linkage indicators are species or groups of species which serve to demonstrate the interdependence of the system components in general, and biogeochemical and socioeconomic spheres in particular. An ideal linkage indicator is one whose response to perturbation is strongly correlated with the response of other indicators. This increases the cost-efficiency of assessment and helps to make assessment socially relevant.

An example of a linkage indicator might be the biochemical response of a particular sports fish to exposure to a particular chemical released at a point source discharge. Knowledge of the species biochemical/physiological response to chemical exposure might allow researchers to predict the population consequences and ultimately the effect on the sports fishing industry in general, of the particular point source discharge. This process firmly places anthropogenic activity in the context of the ecosystem and demonstrates the effect that one type of human activity (i.e., the point source industry) can have on other components of society (the sports fishing industry). Notice that linkage indicators are not necessarily unique but tend to be indicators already developed for other roles but which are also highly correlated with one (or more) other indicators (IJC 1991).

Thus, the selection of an appropriate indicator species is contingent upon the specific role that indicator is to play and at the scale (spatial or temporal) at which it is to be employed. An ideal biomonitoring program would probably include several indicator species in each role. In the case of early warning indicators and diagnostic indicators, an even larger suite of indicators might be required. One of the most important consequences of an approach that explicitly identifies the specific function of each potential indicator is the use of specific management objectives to dictate indicator selection and the resulting logical link between indicator species and management decisions. In all cases the role or function of the particular bioindicator should be explicitly stated.

3.4.2 Types of Bioindicators

It is not the purpose of this report to provide a complete review of biomonitoring techniques employing individuals and populations. Nor would it be possible to adequately review such a very large and growing literature within the confines of this project. Rather, a brief overview of the different response variables used in biomonitoring at the level of the individual organism and population will be provided. References given in the text will provide a starting point for readers interested in more fully exploring certain specific techniques.

The use of biochemical indicators is a relatively new and rapidly growing field within aquatic biomonitoring. Because most responses to stress will first manifest themselves at this level, biochemical processes should be sensitive indicators of individual health and should also serve as both diagnostic and early warning indicators of exposure to stress. However, evidence of exposure, does not of itself provide evidence of ecological effect or consequence. It is important therefore, to develop an improved understanding of the relationship between change in biochemical processes and their effects on individual survival, performance, behaviour, growth and fecundity.

Within the area of biochemical indicators considerable attention is currently devoted to the response of fishes exposed to chemical stressors. In Canada, much of the research in this area has been conducted by K. Munkittrick (Munkittrick and Dixon 1989; Munkittrick *et al.* 1991, 1992).

This work has concentrated around studies of certain fish (notably lake whitefish *Coregonus clupeaformis* and white sucker *Catostomus commersoni*) whose hepatic mixed-function oxygenase (MFO) activity and plasma sex steroid levels have been developed as indicators of exposure to bleached kraft pulp mill effluent. In addition to MFO induction and measures of sex steroid levels, biochemical indicators are currently being developed for energy metabolism; enzyme activity; RNA, DNA, amino acid and protein content and ion regulation (Cormier and Racine 1992; Johnson *et al.* 1993).

Although many biochemical indicators show great promise, most (particularly those dealing with macroinvertebrates) are not yet sufficiently well developed to be of use in routine aquatic biomonitoring programs (Jenkins and Sanders 1992; Johnson *et al.* 1993). Once fully developed these techniques will likely be of more use in diagnosing specific problems rather than as part of a routine basin-wide monitoring program.

Physiological indicators include measurements of change in respiration rate and scope for growth (Seager *et al.* 1992). Traditionally, many of the studies of toxicant effects on respiration have been confined to laboratory tests and this approach continues to be a valued tool in the assessment of chemical hazard. However, there is a growing awareness of the need to assess the risk as well as hazard of chemicals or combinations of chemicals, and such assessments require the use of more ecologically realistic tests. The assessment of risk would involve more realistic laboratory tests, the possible use of mesocosms, and field tests, particularly in combination with assessments of community structure (Seager *et al.* 1992).

As was the case with biochemical indicators, more work is required before physiological indicators can be routinely incorporated in routine aquatic biomonitoring. However, recent advances in the use of telemetry technology already permits the development of physiological indicators in larger organisms such as fish (Seager *et al.* 1992) and may eventually prove feasible in other taxa as well.

Morphological deformities can serve as important indicators of the presence of contaminants and/or other ecological stresses in periphyton (IJC. 1991), invertebrates (Warwick 1989,1991), and fish (Karr 1991, 1992). Measures of morphological deformities have the advantage of being relatively easy to identify and serve as indicators that are of direct and immediate concern to many in the general public (eg. lesions on fish). Unfortunately, such measures also tend to be qualitative rather than quantitative and, in the case of invertebrates, are often taxonomically restricted (Johnson *et al.* 1993). In some instances it may also be difficult to determine the relationship between morphological deformities and ecological endpoints such as survival, growth, reproduction, and performance. Future research exploring the potential of morphological deformity as a bioindicator should concentrate on quantifying deformities and delineating a dose-response relationship between various stresses and the extent or nature of the deformity (Johnson *et al.* 1993).

Bioassays employing behavioural endpoints are increasingly used to assess the effect of contaminants on the environment. Experiments involving behaviourial indicators range from simple laboratory tests to complex mesocosm and field experiments (Buikema and Voshell 1993). Because behaviour tends to be plastic and has its basis in physiology, behavioural modification could serve as an excellent early warning indicator of stress (Henry and Atchinson 1991). However, specific behaviours, particularly in fish, are often difficult to measure in a reliable, repeatable and quantifiable way. Behaviours such as predator avoidance and foraging success may be sensitive to the stress of concern but are also significantly influenced by additional factors such as the history of the test animals, ambient conditions, and the specific experimental design (Henry and Atchinson 1991). Challenges facing those wishing to develop bioindicators that use behavioural endpoints include improvements in experimental design, extrapolation of laboratory results to field situations and the development of appropriate field tests (Henry and Atchinson 1991).

Life-history traits represent an integration of many biochemical, physiological, and behavioural traits and have a direct bearing on population dynamics and community and ecosystem structure. Life-history traits also represent those endpoints most often assumed to best demonstrate ecological effect and are typically directly represented in stated ecosystem objectives (e.g., the existence of a viable fish population). For these reasons, variation in life-history traits could serve as important indicators of current ecosystem condition, long-tern trends within the ecosystem and as linkage indicators. Measures of life-history traits could also serve as early warning indicators (particularly when used laboratory studies employing the appropriate species) and even as diagnostic indicators.

Measures of life-history traits (e.g., determination of age and size class structure of fish and macroinvertebrate populations) have been, and will continue to be an important component of any biomonitoring program. Future research in this area faces several challenges including: (1) The need to consider the often tremendous variation in growth and reproduction observed in natural populations and how this source of variation can be distinguished from anthropogenically induced changes in life-history end points. (2) The to need determine which species within the community show a life-history-trait response to stress at a temporal scale sufficient to allow managers to intervene in a timely fashion. For example, certain fish species may be important to the ecosystem and deserving of routine monitoring but may respond to stress slowly and with time lags. In these cases populations are also likely to recover slowly from stress and may be severely threatened before the stress is even detected. (3) The need to further address complications associated with the extrapolation of laboratory-based observations of growth, survival and fecundity to field data

It should also be noted that well known phenomenon of trade-offs among life-history traits may complicate their interpretation in the context of stress exposure. For example, animals experiencing stress may choose to divert energy away from reproduction and toward growth (Lessells 1991). Under such a scenario, an individual's growth may appear unaffected or even enhanced by exposure to stress, and if growth were the only endpoint measured then the researcher might conclude that the potential stress has no effect. For this reason it is necessary, wherever possible, to measure all relevant life-history end points so that the total effect on the individual can be determined.

3.5 COMMUNITY AND ECOSYSTEM LEVEL BIOMONITORING

Historically, most of the aquatic biomonitoring techniques that rely on measures of community structure have been developed for benthic macroinvertebrate communities. Exceptions to this trend can be found in techniques that make use of zooplankton (Frost *et al.* 1992) or fish (Karr 1991; Dionne and Karr 1992; Fore *et al.* 1994). Nevertheless, benthic mcaroinvertebrate community structure continues to be the most widely used community level aquatic biomonitoring technique. The reasons for this bias are varied. Benthic macroinvertebrate communities, because they tend to move very little, are representative of the area in which they are collected. The same may not be true of fish or zooplankton communities. Like fish and zooplankton, benthic macroinvertebrate taxonomy is fairly well understood and while the cost of sorting and identifying samples may be relatively expensive, the actual collection of data is relatively simple and inexpensive. Indeed, the effort required to adequately sample an entire fish community, particularly in large northern rivers is likely to prove beyond the capability of any biomonitoring program.

For these reasons many of the examples given below are based on research concerning benthic macroinvertebrates. It is important to note that this does not suggest that macroinvertebrate communities are the best or only level at which community biomonitoring should be performed. Rather, it merely reflects a bias in traditional approaches. Many of the techniques described in this report could be readily modified for use in microbial, algal, zooplankton or fish communities. Ultimately, these techniques could be used to simultaneously measure components of several different communities within the aquatic system.

Biomonitoring techniques described in this section and their relevance to the NRBS are summarized in Appendix A of this report.

3.5.1 Saprobic Indices

Originally developed by Kolkwitz and Marsson (1908, 1909, as cited in Metcalfe-Smith 1994), this system introduced the concept of saprobity, or the degree of pollution, in rivers as a measure of contamination by organic matter (primarily sewage) and the consequent decrease in dissolved oxygen levels (Cairns and Pratt 1993). Under the saprobic system, the individual pollution tolerances of various aquatic species from a variety of trophic levels (eg. bacteria, algae, protozoans, rotifers, benthic macroinvertebrates and fish) are determined or scored based on their presence or absence under defined and measured conditions of water quality. The scores for individual species are then combined so as to produce a group score indicating the level of organic enrichment within the system. Although never common in North America (Cairns and Pratt 1993), two saprobic indices are currently being used in Europe, the Biologically Effective Organic Loading plan (BEOL) in Germany and the Quality Index in the Netherlands (Metcalfe-Smith 1994).

Criticisms of saprobic indices include the fact that: (1) The level of taxonomic resolution is not sufficient in some groups and is too controversial in others. (2) Intensive sampling effort is required to determine the representation of selected species, particularly rare species, at all sites investigated. (3) The species list and saprobic values resulting from an application of this approach are site-specific and not easily transferred to other locations. (4) Pollution tolerances of individual species are not assessed in any quantitative fashion. (5) Because each taxon is considered separately, no information is provided at the level of the community (Jones *et al.* 1981, Slooff 1983; Metcalfe-Smith 1994). Finally, saprobic indices were designed to assess organic pollution in the form of sewage discharges and do not provide information on the effect of other important pollutants such as heavy metals, furans and dioxins (Cairns and Pratt 1993; Metcalfe-Smith 1994).

3.5.2 Diversity Indices

Diversity indices combine measures of species richness (the number of species present at a site), evenness (the degree of uniformity in the distribution of individuals among species) and abundance (the total number of organisms present at a site) as an over-all indication of community-level response to the environment. Within this approach, undisturbed environments are assumed to show high species richness, to display generally higher abundance, and to possess a more even distribution of species. Conversely, polluted environments are thought to experience a marked decrease in both species richness and in evenness and tend to be dominated by fewer, more pollution tolerant species. Although a great many different diversity indices have been developed in the past thirty years the Shannon Diversity Index is by far the best known and most widely used.

Diversity indices were once very popular in North America but are not currently favoured as a biomonitoring tool (Brinkhurst 1993; Cairns and Pratt 1993). Their perceived advantages included the fact that they produced quantitative, dimensionless values that were thought (incorrectly) to be amenable to statistical analyses. Calculated diversity values were also relatively independent of sample size and they differed from the saprobic indices in that they did not rely on the subjective assessment of pollution tolerances of individual species or groups (Metcalfe-Smith 1994).

More recently, researchers have discouraged the use of diversity indices for several reasons, including: (1) they contain no information on individual species, (2) they have a history of being misused by managers, and (3) there exist other statistical techniques which retain more biological information in a more ecologically meaningful form (Green 1979). Diversity indices have also been criticized on the basis that they assume (falsely) that all undisturbed sites will have a higher diversity than will all impacted sites. Low levels of pollution have been shown to increase species abundance without eliminating any species thus giving a false negative under certain circumstances (Metcalfe-Smith 1994). Finally, diversity indices have been shown to be poor discriminators between reference sites and those with very low to moderate levels of pollution (Metcalfe-Smith 1994).

3.5.3 Biotic Indices

Biotic indices combine species diversity measurements within a particular subset of taxonomic groups with a knowledge of the pollution tolerances of individual species, genera or families within that group into a single index or score (Metcalfe-Smith 1994). Biotic indices of this type are most often used in rapid assessment approaches to biomonitoring (Resh and Jackson 1993). Well known and currently used biotic indices include the Trent Biotic Index, the Biological Monitoring Working Party (BMWP) and the Belgian Biotic Index (BBI)(Metcalfe 1989; Resh and Jackson 1993).

The biotic indices approach possess several advantages that make them highly attractive to managers: (1) sampling is not intensive, usually involving one to three samples per year), (2) most indices do not require direct measures of total abundance at a given site, and (3) in the case of benthic macroinvertebrates, sampling can often be performed quickly and easily with a kick sampler or hand net. The effect is to greatly reduce the costs of obtaining and processing samples relative to most other current biomonitoring techniques. Another advantage of this approach is that samples can be readily collected and analyzed by workers possessing only moderate biological training.

Biotic indices have been criticized on the basis that most do not consider habitat differences when applying the technique. In other words, they do not include measures of environmental variables. As with the saprobic indices, tolerances of individual taxa to pollutants are typically determined subjectively rather than quantitatively (Metcalfe-Smith 1994). In addition, the scores resulting from biotic indices approach are often not amenable to statistical investigation (Norris and Georges 1993). It is also important to note that while biotic indices may be adapted to assess other forms of pollution, most have been developed to assess the effect of organic pollution (i.e., sewage) only (Metcalfe-Smith 1994).

Overall biotic indices of the type described here, represent a simple and relatively inexpensive technique allowing for the rapid assessment of organic enrichment (and possibly other types of pollution) on potentially impacted sites (Resh and Jackson 1993). Biotic indices could represent an important "first cut" at the data in any proposed NRBS biomonitoring program. Results of biotic indices application could identify reaches of the river requiring more detailed investigation provided that the important limitations of this approach are explicitly recognized. In addition, because biotic indices tend to be region specific, currently available techniques would have to be modified and adapted for use in the Peace, Athabasca and Slave river systems. Developing biotic indices for these systems could entail a considerable investment of time and effort, and thus may not ultimately prove to be a useful or cost-effective component of any biomonitoring program in this region

3.5.4 Community Comparison Indices

Community comparison indices are designed to compare the similarity in structure, of two communities. This comparison may involve two communities separated spatially or may involve comparisons made within a single community at different times (Resh and Jackson 1993). To date, most research in the development of these indices has focused on terrestrial systems and community indices have not yet been widely applied in aquatic systems (Reynoldson, and Metcalfe-Smith 1992). Among those community comparison indices currently in use the Percent Similarity Index, the Pinkham and Pearson's B, and the Bray-Curtis index are among the best known.

Despite not being widely used in aquatic systems, community comparison indices possess several advantages that are not characteristic of the biomonitoring techniques described above. Although an intensive sampling effort and the clear delineation of "clean" reference and impacted sites are required, community comparison indices, unlike biotic and saprobic indices, are sensitive to most forms of perturbation. They are well suited both to long term monitoring at a single site and to making comparisons between or among impacted and reference sites. This approach may also prove particularly valuable in the monitoring rare species (Metcalfe-Smith 1994). Unfortunately, these indices are also insensitive to certain types of community change (e.g. increases in the abundance of dominant taxa) and are extremely sensitive to sampling errors and the presence or absence of rare species.

Given the current state of development of community comparison indices and their sensitivity to sampling error (thus requiring an intensive sampling effort at each site) it seems unlikely that this approach will be of value to NRBS in the near future. Further developments in research may however, make this approach more relevant and cost-effective to those wishing to monitor aquatic ecosystems.

3.5.5 Functional Feeding Groups

The functional feeding group approach (Merritt and Cummins 1984) to aquatic biomonitoring of benthic macroinvertebrates combines the River Continuum Concept with a knowledge of food acquisition techniques and/or mouthpart morphology of benthic macroinvertebrates. It then makes use of this information to generate predictions as to the presence/absence and distribution of different functional feeding groups within a site. This approach is based on the assumption that as pollution levels change within a site so does the distribution of functional feeding groups. For example, an undisturbed site typified by autotrophically-driven processes might have relatively large numbers of individuals or taxa belonging to the scraper feeding group and relatively few that belong to collector-filter or gatherer functional group. As organic pollution levels increase this trend is reversed and the community would be increasingly dominated by collector-filters and gatherers.

The chief advantage of this approach is its reliance on ecological role of species rather than their taxonomic classification. The functional feeding group approach provides much more relevant ecological information, is broadly applicable to a number of different sites and possess considerable intuitive appeal.

Unfortunately, the functional feeding group approach requires an intensive sampling effort to ensure capture of rare species. In addition, this approach also necessitates taxonomic identification to the level of genus or species so as to ensure proper classification into appropriate functional feeding groups. The use of functional feeding groups as biomonitoring tool is further complicated by the fact that many benthic macroinvertebrates are opportunistic feeders and are thus difficult to classify into functional feeding groups. There is also some debate over whether individual taxa should be classified on the basis of mouthpart morphology and food acquisition techniques or on a more direct analysis of stomach contents (Resh and Jackson 1993). Finally, functional feeding groups, while sensitive to organic pollution, do not appear to be sensitive to other toxicants (e.g. heavy metals) in a predictable manner (Luoma and Carter 1991) thus greatly limiting their utility in a general biomonitoring program.

Although it has definite limitations, the functional feeding group approach directly provides relevant ecological information and like biotic indices could be incorporated into a final biomonitoring program for the NRBS. Fortunately, the types of data collected for these approaches are generally contained within the data and sampling requirements of other approaches described below. If data is carefully and properly collected managers will ultimately have the flexibility of applying any number of community level biomonitoring approaches to the same data set.

3.5.6 Indices of Biotic Integrity

These indices are of two types: the index of Biotic Integrity, or the IBI (Karr 1991; Dionne and Karr 1992; Fore *et al.* 1994) which compares sites on the basis of observed fish community structure, and the more recently developed, Benthic Index of Biotic Integrity, or the B-IBI which measures the benthic macroinvertebrate community at different sites (Karr 1992; Kerans and Karr 1994). In each case the community present at a reference, or control, site is compared to that present at an impacted, or polluted, site. Differences between the reference and polluted sites are used to score the impacted site on a scale ranging from 12 (very poor) to 60 (excellent). The score itself is based on values assigned to up to 13 different metrics which include measures of taxonomic richness, proportion of certain selected taxa, proportion of habitat and/or trophic-specific guilds of species, genera or families of sensitive species, tolerant species, individual condition, and abundance.

Both types of biotic integrity indices possess a number of strengths and over the last decade have come to be widely used in the assessment of a variety of stream and smaller river systems, particularly in the United Sates (Steedman 1988; Hoefs and Boyle 1992; Oberdorff and Hughes 1992; Whittier and Rankin 1992; Kerans and Karr 1994). These indices are sensitive to different forms of perturbations (i.e., they are not restricted to the assessment of organic pollution only) as well as to cumulative impacts, they provide ecologically relevant information in so far as they directly measure resource condition and, although they may require modification to account for regional particularities in fish/benthic macroinvertebrate distribution and community structure, they involve relatively easily calculations.

The use of IBI and B-IBI has been criticised on the basis that, in at least one river system, not all metrics respond as expected in the presence of stress (Hoefs and Boyle 1992). In the case of the benthic index of biotic integrity, even those researchers who have developed the approach suggest further testing is required to evaluate both the attributes measured and the index itself (Kerans and Karr 1994).

Other weaknesses of this approach include the fact that the resulting scores do not relate directly to any observable phenomenon, nor to any theoretical or empirical synthesis (Steedman and Regier 1990; Regier 1992). In addition, because IBIs and B-IBIs are adapted and calibrated to local conditions comparing scored sites from different locations is not possible (Regier 1992). The IBI or B-IBI calculated value has no predictive power. The final calcultaed value can mask important information contained in component scores (eg., uniformly mid-range component scores will gave the same final value as will several very high component scores in combioation with several low component scores). Unlike the multivariate approaches described below these indices do not directly measure environmental variables and are therefore not capable of providing inferences as to the cause of deviations away from expected community structure. In short, while the field surveys conducted to calculate the IBI score produce valuable information, this information is much less useful when rendered down to a single number between 12 and 60.

It should also be noted that indexes measuring biotic integrity (IBI or B-IBI) have yet to be applied or tested in any large river system. With respect to the IBI, logistic difficulties, physical effort and financial costs associated with properly measuring and describing fish community structure, particularly in large northern rivers such as the Peace, Athabasca and Slave river systems, may greatly limit the utility of this approach to the NRBS. Proper identification of impacted and reference fish communities may itself provide a considerable challenge in a system in which individual fish are capable of moving hundreds of river kilometres a year, and spend extended periods both upstream and downstream of particular point source discharges of interest. Age-specific habitat requirements of many fish species will also complicate the successful application of the IBI approach.

As was the case with the functional feeding group approach, the data required to apply the B-IBI approach are contained within the sampling criteria of the multivariate approaches described below. The consequence for the NRBS is that properly collected data can be evaluated in a variety of ways at no extra costs. The strengths and weaknesses of these approaches can then be evaluated within the context of the NRBS and that approach or approaches that best satisfies the management objectives can be pursued in greater detail.

3.5.7 Multivariate Approach (*sensu* RIVPACS)

RIVPACS (the River InVertebrate Prediction and Classification System) and other similar multivariate techniques are based on the combination of a detailed knowledge of aquatic macroinvertebrate community structure (often expressed as a BMWP or ASPT score) at a given site, with physical and chemical data (i.e., environmental variables) collected from the same location (Fruse *et al.* 1984; Wright *et al.* 1984, 1987; Moss *et al.* 1987). Multivariate techniques are then used to determine the relationship between community and environmental data and to make predictions as to the expected structure of the aquatic community at a given site. The resulting model is a robust and powerful indicator of expected community structure and shows very high success (> 70%) in correctly classifying sites among as many as 25 different groupings. This approach is also useful in determining the nature of an expected or "target" community for a given site. Target communities can then serve as goals and indicators of progress in any remediation program.

As with indices of biotic integrity multivariate approaches to the study of community composition are sensitive to a wide variety of pollution types and to the combined effects of multiple pollutants. Unlike the IBI or B-IBI, multivariate approaches can also supply managers with indications of the possible mechanisms responsible for shifts in community structure because they measure both biological, environmental, and physicochemical variables. In other words, these models have predictive value. They cannot, by themselves, determine underlying causal mechanisms but they can provide direction as to the most important factors to investigate using experimental techniques.

The strength of these models rests on access to a large data base of environmental and community data. RIVPACS for example, currently derives predictions based on a data set comprised of 438 sites from 80 different rivers with measurements of up to 28 predictor variables from each site (Reynoldson and Metcalfe-Smith 1992).

Obviously such a data base is not currently available for the Slave, Peace, and Athabasca river basins. Nor has there been any consistent attempt to simultaneously collect the required macroinvertebrate and environmental data to run such a model within these basins. Indeed, the creation of such a data base would represent a considerable (and possibly prohibitive) investment of time, effort and expense. In addition, the accuracy of the model will decrease when it encounters environmental values outside the range present in its data base. This suggests that the information base dealing with expected community structure is not directly transferable from one site to another. In other words, data collected as part of similar programs conducted in other regions may be of little direct value to the NRBS.

There are however, several advantages associated with taking the trouble to develop such a data base: (1) The resulting model of community composition is a robust and powerful indicator of expected community structure. The model is sensitive to all types of perturbation and contains explicit information on the effects of changing environmental variables on aquatic community structure. (2) Because different biomonitoring techniques typically require similar types of data, a data base established for multivariate analysis of community structure could also be interpreted in light of other biomonitoring applications. This would allow managers to pick and choose from among available biomonitoring techniques to select the one that best fills their immediate objectives. (3) An information-rich data base of this type would also better accommodate advances in biomonitoring research and technique refinement as the required data would most likely already be collected and available. (4) The establishment of such a data base would make a significant contribution to the understanding of the basic ecology of the system (section 2). (5) Although initially developed to measure aquatic macroinvertebrate community structure there is no reason the same techniques could be successfully applied to other communities such as the algal community or to riparian vegetation.

The establishment of such a data base would also fulfil one of the major objectives of the NRBS, namely to "provide baseline information with regard to the Peace, Athabasca and Slave river basins, both to establish current contaminant levels within the aquatic environment and to develop a baseline for future comparisons" (NRBS 1993) and thus directly address concerns identified by stakeholders.

3.5.8 Multivariate Approach *(sensu* BEAST)

Beast (the BEnthic Assessment of SedimenT) (Zarull and Reynoldson 1992; Reynoldson and Zarull 1993; Reynoldson *et al.* 1994) has been developed for use in the Great Lakes region and makes use of an approach essentially similar to that employed by RIVPACS but with several minor modifications in the collection and analysis of data and in the use of important additional procedures involving sediment toxicological testing. Within BEAST, patterns in macroinvertebrate community structure are investigated using ordination and cluster analysis. Results of ordination analysis are then correlated with environmental variables to determine which of the measured environmental variables are most strongly associated with variability in macroinvertebrate community structure. Multiple discriminant analysis is then used to relate site groupings from pattern analysis to the environmental variables and to generate a model which can be used to predict community structure at other sites with unknown but potential contamination but for which environmental data is available.

In addition to collecting information on the structure of biological communities and on a variety of environmental variables, BEAST also includes laboratory-based bioassays which measure the life-history responses (survival, growth, reproduction) of four benthic invertebrates exposed to sediment collected from the same site. Thus this approach provides information both on general community structure and on the life-history traits of selected taxa exposed to sediment collected from the environment.

BEAST possess all the advantages (and disadvantages) of RIVPACS discussed above but goes further in so far as it provides information on community function (bioassays) as well as community structure (multivariate analyses) at each site. In a study of nearshore areas of the Great Lakes, BEAST correctly predicted benthic macroinvertebrate community structure > 86% of the time (Reynoldson *et al.* 1994).

This approach holds a great deal of promise for biomonitoring. Although specifically designed to address the nature of benthic macroinvertebrate community structure and function, the techniques involved could be modified to permit an examination of other components of the ecosystem or indeed, to combine different components into one larger analysis. As with RIVPACS, BEAST takes direct measures of environmental variables including pollutant levels and thus has the potential to provide managers with insight as to the causal mechanisms responsible for observed shifts in community structure. Like RIVPACS, BEAST also requires a considerable initial investment of time and effort in the acquisition and analysis of a large data base, but this initial investment may well prove cost-effective over the long term.

4.0 BIOMONITORING AND NRBS

Biomonitoring programs, particularly those operating on the scale required by the NRBS are both expensive and labour intensive. As in all cases, the development of an effective biomonitoring program is dependent on an explicit statement of management goals and objectives (reflecting stakeholder input) which will provide a framework for the development of a biomonitoring program and a means by which its success can be measured. To assist in the development of such a framework, the Study Board of the NRBS has identified a series of fourteen scientific and two non-scientific questions which are to serve as a guideline to help the Study satisfy its objectives (Appendix B, NRBS 1992). This series of questions have served to guide research conducted within the NRBS to date, and will no doubt form the basis upon which any future biomonitoring program will be developed.

The development of specific ecosystem goals and objectives is essential to the development of an effective biomonitoring program because it provides a framework within which the monitoring program itself would develop. It also represents a process by which all stakeholders (informed by the best available scientific knowledge) determine the nature of the world in which they want to live.

Clearly, this is a societal decision and not a scientific issue. Science plays a role in refining general goals and in developing specific monitoring objectives that will help satisfy those goals, but the goals themselves must first be determined by society. Any biomonitoring program developed solely on scientific priorities could prove unpopular with the public at large and would be very unlikely to receive legislative approval and support. It is strongly recommended that a process be undertaken whereby scientists and stakeholders begin to develop an explicit system of ecosystem goals and objectives to be used in ecosystem management program for the Peace, Athabasca, and Slave River basins.

This process of public consultation and refinement of management goals and objectives is particularly important for the NRBS. In other regions, such as the Great Lakes, similar processes have been undertaken but in most of those cases remediation was the clear and primary management objective and one that was shared by scientists, government, and the general public. Stakeholders provided input as to the desired levels and types of remediation but all participants began the process with the belief that "something had to be done".

In the case of the Peace, Athabasca and Slave river basins there does exist a general concern regarding the health and integrity of these ecosystems but the best available evidence suggests that these river systems are relatively pristine, particularly when compared to other large river systems throughout the world. This condition presents both an opportunity and a difficulty for those attempting to manage these basins. The opportunity is to develop and put in place an ecosystem management strategy whose primary goal is to largely preserve the system rather than undertake large-scale, long-term, and costly remediation efforts that may or may not satisfy the ecosystem goals. In other words, there is an opportunity within these systems to prevent large-scale impacts rather than to institute policies to recover from such impacts. The difficulty refers to the fact that, in the absence of obvious deleterious impacts, the societal will required to develop an effective ecosystem management strategy may be lacking. Indeed, there may exist those groups that feel that the lack of obvious and dramatic impacts is evidence that the system can handle further anthropogenic inputs.

The existence of this opportunity, as well as the challenges associated with it, serves to emphasize the need to undertake a process by which stakeholders, acting on the best available scientific information, begin to formulate general ecosystem goals. These goals must then be refined into more specific management goals and objectives and it is the development of these specific management objectives that will ultimately determine which of the available biomonitoring tools are most effective.

The selection of the most appropriate biomonitoring tools will be dependent not only on the formulation of ecosystem and management goals and objectives but also on the results of studies currently underway within NRBS. These studies are designed to synthesize and further develop our ecological understanding of the Peace, Athabasca and Slave river systems and to assess the applicability of specific biomonitoring techniques to these basins. Ideally, recommendations concerning biomonitoring would only be provided after a careful consideration of these, as yet uncompleted, studies.

A final constraint on the ability to provide specific and realistic biomonitoring recommendations is the lack of any information on the scale of any future biomonitoring program. Under a scenario of unlimited resources, many of the techniques alluded to in this report could be applied and would yield useful information. Unfortunately, limited budgets require the prioritization of options and necessitate limited investments of time and energy. The lack of information on the resources available for any biomonitoring program complicates the process of providing recommendations.

Despite these limitations (lack of ecosystem and management goals, unavailability of results from ongoing research, lack of information as to future commitment to biomonitoring) it is nevertheless possible to draw a few generalizations as to the form any aquatic biomonitoring program might take. It is also possible to provide general recommendations as to what specific types of tools might prove useful to the NRBS.

As discussed earlier in this report, there is a great and urgent need to construct a "quality" data set for the Peace, Athabasca and Slave river basins. Such a data set would contain information not only on the structure and function of the biotic community at a given site but would also characterize the site in terms of physical and chemical variables. It remains to be determined how many sites would require sampling and how often each site should be sampled but it is clear that any basin-wide effort would represent a considerable investment in both time and effort. In the case of the NRBS, the collection of this type of baseline data is an explicit objective of the program (NRBS 1993) and is necessary to the development of any effective biomonitoring program.

It is important that data not be collected merely for its own sake (as has occurred in some areas), rather there is a need for a carefully designed sampling program which will provide the maximum return of information for invested energy. An ideal biomonitoring program would collect data amenable to techniques as divergent as rapid assessment (e.g. biotic indices) and multivariate (e.g. BEAST) analyses. It is worth noting that many of the biomonitoring techniques discussed in this report have similar data requirements. This is particularly true in community level biomonitoring and suggests that costs associated with different techniques, at least with regard to data collection, may not differ greatly. Costs associated with analyzing the data once collected will vary according to the level of taxonomic resolution required and by the types of statistical techniques applied to the data. Costs in these areas could be reduced if a lower taxonomic resolution was found to provide the same quality of information and by the increasing availability of sophisticated statistical software.

Multivariate approaches such as RIVPACS and BEAST clearly provide the maximum amount of useful information. They measure both environmental and biological variables, are sensitive to a wide variety of stressors and have direct predictive value. The cost of establishing an initial data base for these approaches may be considerable but could be minimized if ongoing sampling efforts within these basins were planned and coordinated so as to provide the required data. The initial cost of these techniques would also be offset by a reduction in monitoring costs once the data base is established.

In addition to using multivariate techniques to monitor macroinvertebrate community structure, I would also recommend that these same techniques be used to examine other aquatic or riparian communities including algal and plant communities. Inclusion of other communities will provide additional information and will constitute a move toward biomonitoring at an ecosystem level. Conversely research may demonstrate that monitoring one community (e.g. the algal community) provides all the required information and at a reduced cost. As discussed above, the costs and difficulties associated with the proper sampling of fish communities within large northern rivers are likely to preclude this ecosystem component from being used in community level biomonitoring. This does not suggest that individuals or populations within this community could not be useful a biomonitoring tool.

This report would thus recommend a multivariate approach of the type developed for RIVPACS and BEAST for community level biomonitoring within the NRBS. These multivariate techniques are capable of assessing current ecosystem condition and of providing information on long-term trends within the ecosystem. They also evaluate community structure in the light of environmental data. I would further suggest that this approach be extended to other aquatic and riparian communities, but that fish communities not be included in this approach.

In addition to community level approaches, an effective biomonitoring program should also develop and employ a suite species or population level bioindicators. As discussed above, bioindicator species should be selected from several different trophic levels and/or components of the ecosystem and would fulfil specific predetermined roles such as indicators of ecosystem state, indicators of ecosystem change, early warning indicators, diagnostic indicators, linkage indicators, and indicators of regulatory compliance.

In some cases, only the presence or absence of the bioindicator need be evaluated, while in others individuals must be evaluated in terms of their biochemistry, physiology, morphology, behaviour, or life-history characteristics while populations would be evaluated for growth and size or age class structure. In the case of species level bioindicators, care must be taken in relating the results of toxicity/bioassay tests to the natural environment and in distinguishing between contaminant exposure and ecological consequences. In all cases, the distribution, abundance and general ecology of the candidate species or population must be understood before it can serve as an accurate bioindicator.

Bioindicators that provide measures of life-history endpoints such as survival, growth and reproduction are particularly valuable because they constitute relevant ecological endpoints, represent an integration of all stresses acting on the individual and are measures readily appreciated by stakeholders. Bioindicators of this type may range from laboratory-based bioassays of invertebrates to field measures of fish populations. These indicators provide information on ecosystem state as well as ecosystem change and, if the individuals within these populations show a sufficiently rapid response to stress, may also serve as early warning indicators. Measures of life-history endpoints are thus likely to prove very useful in routine biomonitoring. Biomonitoring techniques providing information regarding life-history endpoints will also directly address many management objectives, for example, the preservation of a healthy and viable sport or game fish population.

Once a potential problem has been identified within the ecosystem bioindicators that make use of biochemical, physiological and/or behavioural endpoints may prove particularly valuable as diagnostic indicators. Many of these techniques involve a laboratory component, they can be costly and require considerable skill to apply successfully. Their use as indicators of general ecosystem condition is limited because the ability to extrapolate from the biochemical to ecosystem scale is fraught with difficulty and because of the logistic difficulties associated with widespread application of these techniques. However, they also represent a measurable, repeatable and rapid response to stress and in many cases may provide direct evidence of the cause of stress itself. For these reasons techniques involving such biochemical/physiological responses as MFO induction will be useful in diagnosing problems once identified and in examining which components of complex contaminants may be responsible for the observed adverse effects on growth, survival and reproduction.

Other approaches, such as those employing semi-permeable membranes to assess contaminant loadings, are being developed within the NRBS and represent a potential link between biochemical indicators and routine monitoring. The ultimate utility of such an approach, however, has yet to be determined.

This report would thus recommend that indicators that measure life-history end points be used whenever possible to assess ecosystem state and long-term trends and that biochemical, physiological, and/or behavioural indicators be used to provide early warnings and diagnosis of problems within the ecosystem. In all cases, there should be an explicit statement of what role the potential indicator plays and how it relates to specific management objectives. It should also be noted that the utility of any given bioindicator will be constrained by a knowledge of its ecology, physiology and/or biochemistry.

In addition to the development of appropriate community, population and individual level biomonitoring techniques, ongoing experimentation should play an important role in any biomonitoring program. Experiments at the scale of laboratory, microcosm, mesocosm, and/or ecosystem can play an important role in validating data collected as part of routine biomonitoring, and represents the only way to test directly hypotheses generated by an examination of monitoring data.

Properly controlled experiments are also capable of revealing underlying causal mechanisms responsible for observed change at the community or ecosystem level and are useful for the identification of potential bioindicators. Experiments may also provide insight into those components of the ecosystem particularly vulnerable to stress and to the appropriate taxonomic level at which to employ biomonitoring techniques.

Finally, as research in aquatic biomonitoring continues to advance there will no doubt be significant improvements in available techniques and theoretical approaches. Unfortunately, it is difficult to predict the nature of these advancements and the direction from which they will come. For these reasons it is important to maintain a flexible aquatic biomonitoring program that will be capable of adapting and incorporating new or improved techniques as they become available.

Once again this process will be greatly facilitated by the careful collection and archiving of high quality data and the development of an historical data base that could be reanalyzed using new techniques.

Literature Cited

Brinkhurst, R. O. 1993. Future directions in freshwater biomonitoring using benthic macroinvertebrates. In: Rosenberg, D. M., and V. H. Resh (eds.), Freshwater Biomonitoring and Benthic Macroinvertebrates. Chapman and Hall, New York.

Buikema, A. L. Jr., and J. R. Voshell Jr. 1993. Toxicity studies using freshwater benthic macroinvertebrates. In: Rosenberg, D. M., and V. H. Resh (eds.), Freshwater Biomonitoring and Benthic Macroinvertebrates. Chapman and Hall, New York.

Burton, G. A. Jr. 1992. Plankton, macrophyte, fish, and amphibian toxicity testing of freshwater sediments. In: Burton, G. A. Jr. (ed.), Sediment Toxicity Assessment. Lewis Publ. Inc., Boca Raton, FL.

Burton, G. A. Jr., M. K. Nelson, and C. G. Ingersoll. 1992. Freshwater benthic toxicity tests. In: Burton, G. A. Jr. (ed.), Sediment Toxicity Assessment. Lewis Publ. Inc., Boca Raton, FL.

Cairns, J. Jr., B. R. Niederlehner, and E. P. Smith. 1992. The emergence of functional attributes as endpoints in ecotoxicology. In: Burton G. A. Jr. (ed.), Sediment Toxicity Assessment. Lewis Publ. Inc., Boca Raton, FL.

Cairns, J. Jr., McCormick, P.V., and B. R. Niederlehner. 1993. A proposed framework for developing indicators of ecosystem health. Hydrobiologia 263: 1-44.

Cairns, J. Jr., and J. R. Pratt. 1993. A history of biological monitoring using benthic macroinvertebrates. In: Rosenberg, D. M., and V. H. Resh (Eds.), Freshwater Biomonitoring and Benthic Macroinvertebrates. Chapman and Hall, New York.

Calow, P. 1993. Can ecosystems be healthy? Critical consideration of concepts. Journal of Aquatic Ecosystem Health 1: 1-5.

Calow, P. (in press) Aquatic ecosystem health? Journal of the North American Benthological Society.

Christie, W. J., M. Becker, J. W. Cowden, and J. R. Vllentyne. 1986. Managing the Great Lakes Basin as a home. Journal of Great Lakes Research 12: 2-17.

Clements, W. H. 1991. Community responses of stream organisms to heavy metals: a review of observational and experimental approaches. In: Newman, M. C., and A. W. McIntosh (eds.), Metal Ecotoxicology: Concepts and Applications. Lewis Publ. Inc., Chelsea, Michigan.

Connell, J. H. 1978. Diversity in tropical rainforests and coral reefs. Science 199: 1302-1310

Cooper, S. D., and L. Barmuta. Field experiments in biomonitoring. In: Rosenberg, D. M., and V. H. Resh (eds.), Freshwater Biomonitoring and Benthic Macroinvertebrates. Chapman and Hall, New York.

Cormier, S. M., and R. M. Racine. 1992. Biomarkers of environmental exposure and multivariate approaches for assessment and monitoring. In: Mckenzie, D. H., D. E. Hyatt, and V. J. Mcdonald (eds). Ecological Indicators. Elsevier Applied Science. London.

Costan, G., N. Bermingham, C. Blaise, and J. F. Ferard. 1993. Potential ecotoxic effects probe (PEEP): a novel index to assess and compare the toxic potential of industrial effluents. Environmental Toxicology and Water Quality. 8:115-140.

Costanza, R. 1992. Toward an operational definition of ecosystem health. In: Costanza, R., B. G. Norton, and B. D. Haskell (eds.). Ecosystem Health: New Goals for Environmental Management Island Press.

Costanza, R., B. G. Norton, and B. D. Haskell (eds.). 1992. Ecosystem Health: New Goals for Environmental Management Island Press.

Cuffney, T. F., M. E. Gurtz, and M. R. Meador. 1993. Methods for collecting benthic invertebrate samples as part of the national water-quality assessment program. U.S. Geological Survey. Open-File Report 93-406. 66 pp.

Cuffney, T. F., M. E. Gurtz, and M. R. Meador. 1993. Guidelines for the process and quality assurance of benthic invertebrate samples collected as part of the national water-quality assessment program. U.S. Geological Survey. Open-File Report 93-407. 80 pp.

Dionne, M., and J. R. Karr. 1992. Ecological monitoring of fish assemblages in Tennessee River Reservoirs. In: Mckenzie, D. H., D. E. Hyatt, and V. J. Mcdonald, (eds.). Ecological Indicators. Elsevier Applied Science. London.

Dixon, P. M., and M. C. Newman. 1991. Analyzing toxicity data using statistical models for time-to-death: an introduction. In: Newman, M. C., and A. W. McIntosh (eds.), Metal Ecotoxicology: Concepts and Applications. Lewis Publ. Inc. Chelsea, Michigan.

Downing, J. A. 1979. Aggregation, transformation, and the design of benthos sampling programs. Journal of the Fisheries Research Board of Canada 36: 1454-1463.

Fore, L. S., J. R. Karr, and Conquest, L. L. 1994. Statistical properties of an index of biological integrity used to evaluate water resources. Canadian Journal of Fisheries and

Aquatic Science 51: 1077-1087.

Fox, J. 1992. The problem of scale in community resource management. Environmental Management. 16:289-297.

Frost, T. M., S. R. Carpenter, and T. K. Kratz. 1992. Choosing ecological indicators: effects of taxonomic aggregation on sensitivity and natural variability. In: Mckenzie, D. H., D. E. Hyatt, and V. J. Mcdonald (eds.). Ecological Indicators. Elsevier Applied Science.

Fruse, M. T., D. Moss, J. F. Wright, and P. D. Armitage. 1984. The influence of seasonal and taxonomic factors on the ordination and classification of running-water sites in Great Britain and on the prediction of their macro-invertebrate communities. Freshwater Biology. 14:257-280.

Gibbons, W. N., M. D. Munn, and M. D. Paine. 1993. Guidelines for monitoring benthos in freshwater environments. Report prepared for Environment Canada, North Vancouver, B.C., by EVS Consultants, North Vancouver, B.C. 81 pp.

Gray, J. S., and K. Jensen. 1993. Feedback monitoring: a new way of protecting the environment. Trends in Ecology and Evolution. 8:267-268.

Green, R. H. 1979. Sampling Design and Statistical Methods for Environmental Biologists. John Wiley, New York.

Haskell, B. D., B. G. Norton, R. Costanza. 1992. What is ecosystem health and why should we worry about it? In: Costanza, R., B. G. Norton, and B. D. Haskell (eds.). Ecosystem Health: New Goals for Environmental Management Island Press.

Henry, M. G., and Atchinson G. J. 1991. Metal effects on fish behaviour - advances in determining the ecological significance of responses. In: Newman, M. C., and A. W. McIntosh (eds), Metal Ecotoxicology: Concepts and Applications. Lewis Publ. Inc., Chelsea, Michigan.

Hoefs, N. J., and T. P. Boyle. 1992. Contribution of fish community metrics to the index of biotic integrity in two Ozark rivers. In: Mckenzie, D. H., D. E. Hyatt, and V. J. Mcdonald, (eds.). Ecological Indicators. Elsevier Applied Science. London.

IJC (International Joint Commission). 1991. A proposed framework for developing indicators of ecosystem health for the Great Lakes Region. Report to the International Joint Commission. 47 pp.

Jackson, D. A. 1993. Multivariate analysis of benthic invertebrate communities: the implication of choosing particular data standardization, measures of association, and ordination methods. Hydrobiologia. 268:9-26.

Jenkins, K. D., and B. M. Sanders. 1992. Monitoring with biomarkers: a multi-tiered framework for evaluating the ecological impacts of contaminants. In: Mckenzie, D. H., D. E., Hyatt, and V. J. Mcdonald, (eds.). Ecological Indicators. Elsevier Applied Science. London.

Johnson, R. K., T. Wiederholm, and D. M. Rosenberg. 1993. Freshwater biomonitoring using individual organisms, populations, and species assemblages of benthic macroinvertebrates. In: Rosenberg, D. M., and V. H. Resh (eds.), Freshwater Biomonitoring and Benthic Macroinvertebrates. Chapman and Hall, New York.

Karr, J. R. 1991. Biological integrity: a long-neglected aspect of water resource management. Ecological Applications. 1:66-84.

Karr, J. R. 1992. Indices of biotic integrity. In: Marmorek, D. R., T. M. Berry, P. Bunnel, D. P. Bernard, W. A. Kurz, C. L. Murray, K. Paulig, and L. Sully. 1992. Towards an ecosystem approach in British Columbia: Results of a workshop on ecosystem goals objectives, December 7 to 9, 1992. Prepared by ESA Ltd., Vancouver B.C. for Environment Canada, British Columbia Ministry of Environment, Lands and Parks, and British Columbia ministry of Forests. Fraser River Action Plan 193-16. 77 pp.

Kerans, B. L., and J. R. Karr. 1994. A benthic index of biotic integrity (B-IBI) for rivers of the Tennessee Valley. Ecological Applications 4: 768-785.

La Point, T. L., and J. F. Fairchild. 1992. Evaluation of sediment contaminant toxicity: the use of freshwater community structure. In: Burton, G. A. Jr. (ed.), Sediment Toxicity Assessment. Lewis Publ. Inc., Boca Raton, FL.

Lee, B. J., H. A. Reiger, and D. J. Rapport. 1982. Ten ecosystem approaches to the planning and management of the Great Lakes. Journal of Great Lakes Research 8: 505-519.

Lessells, C. A. 1991. The evolution of life-histories. In: Krebs, J. R., and N. B. Davies (eds.), Behavioural Ecology: An Evolutionary Approach Blackwell Scientific Publications, Oxford.

Levin, S. A. 1992. The problem of pattern and scale in ecology. Ecology. 73:1943-1967.

Luoma, S. N., and J. L. Carter. 1991. Effects of trace metals on aquatic benthos. In: Newman, M. C., and A. W. McIntosh (eds.), Metal Ecotoxicology: Concepts and Applications. Lewis Publ. Inc., Chelsea, Michigan.

Marmorek, D. R., T. M. Berry, P. Bunnel, D. P. Bernard, W. A. Kurz, C. L. Murray, K. Paulig, and L. Sully. 1992. Towards an ecosystem approach in British Columbia: Results of a workshop on ecosystem goals objectives, December 7 to 9, 1992. Prepared by ESA Ltd., Vancouver B.C. for Environment Canada, British Columbia Ministry of Environment, Lands and Parks, and British Columbia ministry of Forests. Fraser River Action Plan 193-16. 77 pp.

McBride, G. B., J. C. Loftis, and N. C. Adkins. 1993. What do significance tests really tell us about the environment? Environmental Management. 17:423-432.

Meador, M. R., T. F. Cuffney and M. E. Gurtz. 1993. Methods for sampling fish communities as part of the national water-quality assessment program. U.S. Geological Survey. Open-File Report 93-404. 40 pp.

Meador, M. R., C. R. Hupp, T. F. Cuffney and M. E. Gurtz. 1993. Methods for characterizing stream habitat as part of the national water-quality assessment program. U.S. Geological Survey. Open-File Report 93-408. 48 pp.

Merrit, J. L., and K. W. Cummins (eds.). 1984. An introduction to the aquatic insects of North America, 2nd ed. Kendall/Hunt, Dubuque, IA.

Metcalfe, J. L. 1989. Biological water quality assessment of running waters based on macroinvertebrate communities: history and present status in Europe. Environmental Pollution. 60:101- 139.

Metcalfe-Smith, J. L. 1994. Biological water quality assessment of rivers: use of macroinvertebrate communities, p. 144-170. In: Calow, P., and G. E. Petts (eds.), Rivers Handbook, Volume II. Blackwell, Oxford.

Moss, D., M. T. Furse, J. F. Wright, and P. D. Armitage. 1987. The prediction of the macro-invertebrate fauna of unpolluted running-water sites in Great Britain using environmental data. Freshwater Biology. 17:41-52.

Munkittrick, K. R., and D. G. Dixon. 1989. Use of white sucker (*Catostomus commersoni*) populations to assess the health of aquatic ecosystems exposed to low-level contaminant stress. Canadian Journal of Fisheries and Aquatic Sciences. 46:1455- 1462.

Munkittrick, K. R., M. E. McMaster, C. B. Portt, G. J. Van Der Kraak, I. R. Smith, and D. G. Dixon. 1992. Changes in maturity, plasma sex steroid levels, hepatic mixed-function oxygenase activity, and the presence of external lesions in lake whitefish (*Coregonus clupeaformis*) exposed to bleached kraft mill effluent. Canadian Journal of Fisheries and Aquatic Sciences. 49:1560- 1569.

Munkittrick, K. R., C. B. Portt, G. J. Van Der Kraak, I. R. Smith, and D. A. Rokosh. 1991. Impact of bleached kraft mill effluent on population characteristics, liver MFO activity and serum steroid levels of a Lake Superior white sucker (*Catostomus commersoni*) population. Canadian Journal of Fisheries and Aquatic Sciences. 48:1371-1380.

National Research Council. 1991. Animals As Sentinels of Environmental Health Hazards. National Academy of Science Press, Washington, D.C.

Norris, R. H., and A. Georges. 1993. Analysis and interpretation of benthic macroinvertebrate surveys. In: Rosenberg, D.M., and V. H. Resh (eds.), Freshwater Biomonitoring and Benthic Macroinvertebrates. Chapman and Hall, New York.

Norris, R. H., and T. B. Reynoldson. 1993. Descriptive multivariate analysis: association of benthic invertebrates with sediment quality in the Great Lakes. In: Use of Biostatistics in Benthic Ecological Studies. 6th NABS Tech. Info. Workshop.

Norther River Basins Study. 1992. Northern Rivers Basins Study, Guiding Questions: What the Study Will Be Doing to Answer Them. Prepared by the Northern River Basins Study Office. 8 pp.

Northern River Basins 1993. Annual Report. Prepared by the Northern River Basins Study Office. 32 pp.

Oberdorff, T., and R. M. Hughes. 1992. Modification of an index of biotic integrity based on fish assemblages to characterize rivers of the Seine Basin, France. Hydrobiologia. 228:117-130.

O'Neill, R. V., A. R. Jackson, and A. W. King. 1989. A hierarchical framework for the analysis of scale. Landscape Ecology 3: 193-205.

O'Neill, R. V., C. T. Hausaker, and D. A. Levine. 1992. Monitoring challenges and innovative ideas. In: Mckenzie, D. H., D. E. Hyatt, and V. J. Mcdonald, (eds.). Ecological Indicators. Elsevier Applied Science. London.

Peakall, D. 1992. Animal Biomarkers As Pollution Indicators. Chapman and Hall, New York.

Porter, S. D., T. F. Cuffney, M. E. Gurtz and M. R. Meador. 1993. Methods for collection of algal samples as part of the national water-quality assessment program. U.S. Geological Survey. Open-File Report 93-409. 39 pp.

Rapport, D. J. 1992a. Evaluating ecosystem health. Journal of aquatic Ecosystem Health 1: 15-24.

Rapport, D. J. 1992b. Evolution of indicators of ecosystem health. In: Mckenzie, D. H., D. E. Hyatt, and V. J. Mcdonald, (eds.). Ecological Indicators. Elsevier Applied Science. London.

Rapport, D. J. 1992c. What is clinical ecology. In: Costanza, R., B. G. Norton, and B. D. Haskell (eds.). Ecosystem Health: New Goals for Environmental Management Island Press.

Rapport, D. J., H. A. Reiger, and T. C. Hutchinson. 1985. Ecosystem behavior under stress. American Naturalist 125: 617-640.

Regier, H. A. 1992. Indicators of ecosystem integrity. In: Mckenzie, D. H., D. E. Hyatt, and V. J. Mcdonald, (eds.). Ecological Indicators. Elsevier Applied Science. London.

Resh, V. H., and J. K. Jackson. 1993. Rapid assessment approaches to biomonitoring using benthic macroinvertebrates. In: Rosenberg, D. M., and V. H. Resh (eds.), Freshwater Biomonitoring and Benthic Macroinvertebrates. Chapman and Hall, New York.

Resh, V. H., A. V. Brown, A. P. Covich, M. E. Gurtz, H. W. Li, G. W. Marshall, S. R. Rice, A. L. Sheldon, J. B. Wallace, and R. C. Wissmar. 1988. The role of disturbance in stream ecology. Journal of the North American Benthological Society 7: 433-455.

Resh, V. H., and E. P. McElravy. 1993. Contemporary quantitative approaches to biomonitoring using benthic macroinvertebrates. In: Rosenberg, D. M., and V. H. Resh (eds.), Freshwater Biomonitoring and Benthic Macroinvertebrates. Chapman and Hall, New York.

Reynoldson, T. B., K. E. Day, and R. H. Norris. 1994. Biological guidelines for freshwater sediment based on **BE**nthic Assessment of Sedimen**T** (the **BEAST**) using a multivariate approach for predicting biological state. Draft Report. National Water Research Institute, Environment Canada, Burlington, ON.

Reynoldson, T. B., and J. L. Metcalfe-Smith. 1992. An overview of the assessment of aquatic ecosystem health using benthic invertebrates. Journal of Aquatic Ecosystem Health. 1:295-308.

Reynoldson, T. B., and M. A. Zarull. 1993. An approach to the development of biological sediment guidelines. In: Woodley, S. J., G. Francis, and J. Kay (eds.), Ecological Integrity and the Management of Ecosystems. St. Lucie Press, Delray Beach, FL.

Rose, K. A., and E. P. Smith. 1992. Experimental design: the neglected aspect of environmental monitoring. Environmental Management. 16:691-700.

Rosenberg, D. M., and V. H. Resh. 1993. Introduction to freshwater biomonitoring and benthic macroinvertebrates. In: Rosenberg, D. M., V. H. Resh (eds.), Freshwater Biomonitoring and Benthic Macroinvertebrates. Chapman and Hall, New York.

Schindler, D. W. 1987. Detecting ecosystem responses to anthropogenic stress. Canadian Journal of Fisheries and Aquatic Sciences. 44(Suppl. 1):6-25.

Schindler, D. W. 1988. Experimental studies of chemical stressors on whole lake ecosystems. Internationale Vereinigung fur Theoretische und Angewandte Limnologie Verhandlungen. 23:11-41.

Schindler, D. W. 1990. Experimental perturbations of whole lakes as tests of hypotheses concerning ecosystem structure and function. Oikos. 57:25-41.

Seager, J., I. Milne, and M. Crane. 1992. The application of in situ bioassays as ecological indicators for assessing river quality. In: Mckenzie, D. H., D. E. Hyatt, and V. J. Mcdonald, (eds.). Ecological Indicators. Elsevier Applied Science. London.

Sloof, W. 1983. Benthic macroinvertebrates and water quality assessment: Some toxicological considerations. Aquatic Toxicology 4: 73-82.

Sousa, W. P. 1979. Disturbance in marine intertidal boulder fields: the nonequilibrium maintenance of species diversity. Ecology 60: 1225-1239.

Steedman, R. J. 1988. Modification and assessment of an index of biotic integrity to quantify stream quality in southern Ontario. Canadian Journal of Fisheries and Aquatic Sciences. 45:492-501.

Steedman. R. J., and H. A. Regier. 1990. Ecological basis for an understanding of ecosystem integrity in the Great Lakes Basin. In: C. J. Edwards, and H. A. Regier, (eds.). An ecosystem approach to the integrity of the Great Lakes Basin in turbulent times. IJC Special Publication.

Warwick, W. F. 1989. Morphological deformities in larvae of *Procladius* Skuse (Diptera: Chironimdae): and their biomonitoring potential. Canadian Journal of Fisheries and Aquatic Sciences 46: 1255-1271.

Warwick, W. F. 1991. Indexing deformities in ligulae and antennae of *Procladius* larvae (Diptera: Chironimade): application to contaminant-stressed environments. Canadian Journal of Fisheries and Aquatic Sciences 48: 1151-1166.

Whittier, T. R., and E. T. Rankin. 1992. Regional patterns in three biological indicators of steam condition in Ohio. In: Mckenzie, D. H., D. E. Hyatt, and V. J. Mcdonald, (eds.).

Ecological Indicators. Elsevier Applied Science. London.

Wright, J. F., P. D. Armitage, M. T. Furse, and D. Moss. 1988. A new approach to the biological surveillance of river quality using macroinvertebrates. Internationale Vereinigung fur Theoretische und Angewandte Limnologie Verhandlungen. 23:1548-1552.

Wright, J. F., D. Moss, P. D. Armitage, and M. T. Furse. 1984. A preliminary classification of running-water sites in Great Britain based on macro-invertebrate species and the prediction of community type using environmental data. Freshwater Biology. 14:

Zarull, M. A., and T. B. Reynoldson. 1992. A management strategy for contaminated sediments: assessment and remediation. Water Pollution Research Journal of Canada. 27:871-882.

Appendix A

Overview of Common Community Level Biomonitoring Approaches

Saprobic Indices

Description

Developed by Kolkwitz and Marsson (1908,1909) this system makes use of the presence or absence of indicator species to determine the extent of organic pollution and resulting dissolved oxygen levels.

Pollution tolerances of various species (bacteria, algae, protozoans, rotifers as well as some benthic macroinvertebrates and fish) are determined and combined to give a score indicating the level of organic enrichment.

Two systems are currently in use: Biologically Effective Organic Loading (BEOL) in West Germany and Quality Index (Netherlands).

Data Requirements

- Species level identification.
- Intensive sampling effort.

Strengths

- Provides information at the species level.

Weaknesses

- Requires detailed taxonomic knowledge/keys.
- Catalogues of species sensitivity are developed for each area studied and the information may not be transferable from one location to another.
- Provides information only on organic enrichment.
- Provides no information at community or functional level.

Relationship to Current EEM Guidelines

Is dependent on specific knowledge of pollution tolerances of selected taxa within the northern river basins. Such knowledge would have to be available before saprobic indices could be applied.

Applicability to NRBS

This index is seldom employed in current monitoring programs and is unlikely to be a useful technique for NRBS.

Diversity Indices

Description

Uses measures of species richness, evenness, and total abundance as an indication of the community response to the environment. Undisturbed environments are assumed to be species rich, possess generally higher abundance, and have an even distribution of species. Polluted environments show a drop in diversity and evenness and are dominated by fewer, more tolerant, species.

Although a great many diversity indices have been developed the Shannon Index is by far the best known and the most often used.

Data Requirements

- Species level identification.
- Intensive sampling effort.

Strengths

- Quantitative results are amenable to statistical analyses.
- Results are largely independent of sample size.
- Does not rely on subjective assessments of pollution tolerance of individual species.

Weaknesses

- Indices contain no information on individual species, only numbers.
- Makes assumption that all undisturbed communities possess higher diversity.
- Pollution may not always act to reduce diversity and evenness thus, giving false results.
- Can be insensitive to differences among sites.

Relationship to Current EEM Guidelines

Could be conducted within current framework provided all species in a site were identified.

Applicability to NRBS

Although once widely used, diversity indices are no longer a favoured biomonitoring technique (except in combination with data regarding pollution tolerances of key taxa, see below) and are unlikely to be a useful technique for NRBS.

Biotic Indices

Description

Combines diversity of a subset of taxonomic groups with a knowledge of the pollution tolerances of those groups to generate a single index.

Well known biotic indices include the Trent Biotic Index, the Biological Monitoring Working Party (BMWP) and the Belgian Biotic index (BBI)

Data Requirements

- Usually requires identification only to level of Family.
- Sampling often performed with a hand net or kick sampler.
- Most indices do not require measures of total abundance.
- No intensive sampling effort (often one sample/year).

Strengths

- Data collection is inexpensive.
- Samples can be collected and analyzed by workers with only moderate biological training.
- Useful approach to Rapid Assessment, or in preliminary studies.

Weaknesses

- Most biotic indices do not consider differences in habitat.
- Organism tolerance is often subjectively determined.
- Tolerances of NRBS taxa may not be known.
- Provides information on organic enrichment only.

Relationship to Current EEM Guidelines

Could be conducted within current framework.

Applicability to NRBS

Biotic indices may prove a simple and relatively inexpensive technique allowing for rapid assessment of organic pollution impact on sites. Results of rapid assessment techniques could provide information as to which reaches of the river require more detailed investigations.

Community Comparison Indices

Description

Can be used to compare the similarity in structure of two communities in space or in time. Most research thus far has been focused on terrestrial systems and community indices have yet to be widely applied in aquatic systems. Among those community indices currently in use the Percent Similarity Index , the Pinkham and Pearson's B, and the Bray-Curtis index are amongst the best known.

Data Requirements

- Species level identification.
- In the case of spatial comparisons a clean water reference site (upstream-downstream).
- Intensive sampling effort (in order to adequately sample rare species).

Strengths

- Sensitive to most forms of perturbation (not sensitive only to organic enrichment).
- Can be used to monitor long-term trends at one site or to compare impacted site to a reference site.
- May be useful when preservation of rare species particularly important.

Weaknesses

- Not yet widely used in aquatic systems (may require significant effort to develop program)
- Is insensitive to certain types of community changes (e.g. increase in abundance of dominant taxa).
- Very sensitive to sampling error.

Relationship to Current EEM Guidelines

Cannot be conducted within current framework.

Applicability to NRBS

Community indices may provide insight into community change, however the cost of developing and implementing such a program under the NRBS could prove considerable.

Functional Feeding Groups

Description

The functional feeding group approach combines the River Continuum Concept with a knowledge of food acquisition techniques and/or mouthpart morphology of benthic macroinvertebrates to make predictions as to the distribution of different feeding groups within a site. As pollution levels change within a site so does the distribution of functional feeding groups. For example, an undisturbed site typified by autotrophically-driven processes might have numbers of scrapers relative to collector-filters and gathers. As organic pollution levels increase this trend is reversed.

Data Requirements

- Genus/species level identification (to allow for proper classification into functional feeding group).
- Intensive sampling effort (in order to adequately sample rare species).

Strengths

- Reliance on ecological role rather than taxonomic classification constitutes an approach which provides more relevant ecological information.
- Reliance on ecological role rather than taxonomic classification means that approach need not be modified on a site by site basis to account for local differences in community structure.

Weaknesses

- Requires detailed taxonomic identification.
- Ultimately sensitive only to organic enrichment, insensitive to other toxicants.
- There is debate over whether individual taxa should be classified on the basis of mouthpart morphology and food acquisition techniques or on analyses of stomach contents.
- Many macroinvertebrates are omnivorous and opportunistic and are consequently difficult to classify into functional groups.

Relationship to Current EEM Guidelines

The functional feeding group approach could be incorporated within a slightly modified set of EEM guidelines.

Applicability to NRBS

A knowledge of the functional feeding groups present at any sight would prove valuable. Awareness of the ecological function of selected taxa is also incorporated in the Index of Biological Integrity.

Indices of Biotic Integrity

Description

These indices are of two types: the fish Index of Biotic Integrity (IBI) and the benthic Index of Biotic Integrity (B-IBI). In each case the community present at a reference, or control, site is compared to that present at an impacted, or polluted, site. Differences between the sites are used to score the impacted site on a scale from 12 (very poor) to 60 (excellent). The score itself is based on values assigned to 12 metrics which include measures of taxonomic richness, habitat and trophic-specific guilds of species, sensitive species, tolerant species, individual condition, and abundance..

Data Requirements

- Identification of taxa to the level of species (IBI) or family/genus (B-IBI).

Strengths

- This approach is sensitive to different forms of perturbation and to cumulative impacts.
- These indices provide ecologically relevant information in so far as they directly measure resource condition.
- Measures of biotic integrity are easily calculated.

Weaknesses

- This approach relies on the existence of minimally impacted reference sites of similar size from the same ecological region. Such sites may not be available.
- Metrics may have to be modified to account for regional particularities in fish distribution and community structure.
- Unlike multivariate techniques these indices do not measure environment variables and are therefore not capable of providing direct evidence as to the cause of deviations away from the expected community structure.
- IBI and B-IBI have been criticized because the final number or value contains little information.

Relationship to Current EEM Guidelines

Could be conducted within current framework.

Applicability to NRBS

The data required to calculate integrity indices would be easily collected as part of a more ambitious monitoring program (e.g. the multivariate approach) and measures of integrity could be useful in providing initial assessments of sites. However the value of such an approach will ultimately depend on the availability and proper selection of appropriate reference sites.

Multivariate Approach
(sensu RIVPACS)

Description

This technique, and others like it, combines a detailed knowledge of community structure (sometimes expressed as a BMWP or ASPT score) with physical and chemical data collected from the same locations. Multivariate techniques are used to determine the relationship between community and environmental data and to then make predictions as to the expected structure of aquatic communities based on environmental data alone.

This technique shows high success (> 70%) in correctly classifying sites among as many as 25 different groupings and is very useful in determining an expected or "target" community for a given site.

Data Requirements

- Requires a data base comprised of information (both biological and environmental) collected over a period of several years and (in some cases) from spring summer and fall collections.
- Taxa can be identified to the level of either species or family.

Strengths

- Resulting model is a robust and powerful indicator of expected community structure at a given site.
- Differences between observed and expected community structure provide clear evidence of perturbation and, in some cases, reveal underlying causes.
- Expected community structure can be used as a model or "target" community which remedial actions can attempt to recreate.
- The model is sensitive to all types of pollution and to the combined effects of more than one pollutant.

Weaknesses

- The strength of the model rests on access to a large data base of environmental and community data (e.g. RIVPACS currently derives predictions based on a data set comprised of 38 sites from 80 different rivers and measurements of up to 28 predictor variables from each site).

Weaknesses (continued)

- In order to properly distinguish among different types of communities data should be available from a large number of different reference sites.
- The accuracy of the model will be reduced when the model encounters environmental data which falls outside the range of that present in the data base, thus complicating the application of a model developed in one geographic region to other regions.

Relationship to Current EEM Guidelines

Can not be conducted within current framework

Applicability to NRBS

Techniques such as RIVPACS hold a great deal of promise for biomonitoring. Their applicability to NRBS will however be a function of the quality and extent of the data base available. The generation of such a data base represents a significant investment in terms of time, effort and expense but will ultimately provide a wealth of information. Data collected in the course of this program are also sufficient for several other rapid assessment biomonitoring techniques (e.g. biotic indices, B-IBI).

Multivariate Approach
(sensu BEAST)

Description

This technique is similar to that employed by RIVPACS but differs in several aspects. The approach does not rely on biotic indices but rather classifies community assemblages using ordination and classification techniques. Results of the ordination analyses are then correlated with environmental variables to determine which environmental variables are most strongly associated with variability in community structure. As with RIVPACS this approach allows for the prediction of community structure based on a limited set of environmental variables but in addition employs laboratory based bioassays to assess the life-history responses of selected invertebrate taxa to exposure to sediment.

This technique shows high success (> 86%) in correctly classifying sites among and has the potential to identify specific contaminants which can be used to discriminate among community types. This technique can also be used to describe an expected or "target" community for a given site.

Data Requirements

- Requires a data base comprised of information (both biological and environmental) collected from control (or reference) sites as well as impacted (or polluted sites).
- Taxa are typically identified to the level of family and in some cases genus or species.
- The development of appropriate bioassay techniques and the collection of bioindicators.

Strengths

- Resulting model is a robust and powerful indicator of expected community structure at a given site.
- Provides information not only on environmental variables, including the extent of sediment contamination at a give site, but also provides data on both the structure and function of the community.
- Differences between observed and expected community structure provide clear evidence of perturbation and, in some cases, reveal underlying causes.
- Expected community structure can be used as a model or "target" community which remedial actions can attempt to recreate.
- The model is sensitive to all types of pollution ant to the combined effects of more than one pollutant.

Weaknesses

- As with RIVPACS the strength of the model is dependent on the ability to create a sufficiently large and representative data base.
- In order to properly distinguish among different types of communities, data should be available from a large number of different reference sites.
- The accuracy of this approach model will be reduced when the model encounters environmental data which falls outside the range of that present in the data base, thus complicating the application of a model developed in one geographic region to other regions.

Relationship to Current EEM Guidelines

Can not be conducted within current framework

Applicability to NRBS

This techniques holds a great deal of promise for biomonitoring. Its applicability to NRBS will however be a function of the quality and extent of the data base available. The generation of such a data base represents a significant investment in terms of time, effort and expense but will ultimately provide a wealth of information, not only on community structure but on a variety of environmental variables, including contaminant levels. Data collected in the course of this program are also sufficient for several other rapid assessment biomonitoring techniques (e.g. biotic indices, B-IBI).

Reduced Assemblages

Description

This approach is predicated on the assumption that the information contained within an analyses of the entire community is equally available from an analyses of subset of the same community.

Data Requirements

- Species level identification within the reduced assemblage.
- Intensive sampling effort (in order to adequately sample rare and/or indicator species).

Strengths

- Collection, sorting and identification of reduced assemblage may be less expensive and logistically simpler than similar procedures carried out on the entire community.
- Examination of a reduced assemblage may facilitate greater taxonomic precision.
- Species level identification may allow for the development of bio-indicators.
- Individual species within a reduced assemblage will be more sensitive to perturbation than will higher taxonomic groupings such as genus and family employed in other biomonitoring techniques.

Weaknesses

- Researchers must ensure that examination of some reduced assemblage is capable of adequately reflecting changes in the community as a whole (will require preliminary studies).
- Not capable of detecting loss of species that are not members of the assemblage.
- Inherit site-specific differences in taxonomic composition of reduced assemblages may complicate the application of such an approach within the NRBS.

Relationship to Current EEM Guidelines

Can not be conducted within current framework

Applicability to NRBS

May ultimately prove valuable to NRBS and could include long term savings in cost and time. However, care must be taken to ensure that studies of a reduced assemblage can be used to accurately predict status of entire community.

Ratio Indices

Description

Based on the assumption that the dominance of one taxonomic or functional feeding group over another provides valuable ecological data.

Data Requirements

- Dependent on the particular ratios of interest.

Strengths

- Easily calculated.
- Owing to a concentration on particularly sensitive groups may be more sensitive than other indices in detecting certain types of change.
- Ratios may directly provide important ecological information.

Weaknesses

- As with reduced assemblies researcher must ensure that taxa or groups which form the bases of ratios provide accurate information on changes in the community as a whole.
- Not capable of detecting loss of species that are not used in calculating the ratios.

Relationship to Current EEM Guidelines

Can be conducted within current framework

Applicability to NRBS

The ease with which ratio indices can be calculated may make them useful to NRBS. However ratio indices should only be used in conjunction with other biomonitoring techniques

APPENDIX B

Questions identified by the NRBS Study Board which are to serve as guidelines to help the study meet its objectives (NRBS 1992).

Scientific Questions

1) a) How has the aquatic ecosystem, including fish and/or other aquatic organisms, been affected by exposure to organochlorines or other toxic compounds?
   b) How can the ecosystem be protected from the effects of these compounds?

2) What is the current state of water quality in the Peace, Athabasca and Slave river basins, including the Peace-Athabasca delta?

3) Who are the stakeholders and what are the consumptive and non-consumptive uses of water resources in the river basin?

4) a) What are the contents and nature of the contaminants entering the system and what is their distribution and toxicity in the aquatic ecosystem with particular reference to water, sediments and biota?
   b) Are toxins such as dioxins, furans, mercury, etc. increasing or decreasing and what is their rate of change?

5) Are the substances added to the rivers by natural and man made discharge likely to cause deterioration of the water quality?

6) What is the distribution and movement of fish species in the watersheds of the Peace, Athabasca and Slave rivers? Where and when are they most likely to be exposed to changes in water quality and where are their most important habitats?

7) What concentrations of dissolved oxygen are required seasonally to protect the various life stages of fish, and what factors control dissolved oxygen in the rivers?

8) Recognizing that people drink water and eat fish from these systems, what is the current concentration of contaminants in water and edible fish tissue and how are these levels changing through time and by location?

9) Are fish tainted in these waters and, if so, what is the source of the tainting compounds (i.e. compounds affecting how fish taste and smell to humans)?

10) How does and how could river flow regulation impact the aquatic ecosystem?

11) Have the riparian vegetation, riparian wildlife and domestic livestock in the river basins been affected by exposure to organochlorines or other toxic compounds?

12) What native traditional knowledge exists to enhance the physical science studies in all areas of the enquiry?

13) a) What predictive tools are required to determine the cumulative effects of man made discharges on the water and aquatic environment?

b) What are the cumulative effects of man made discharge on the water and aquatic environment?

14) What long term monitoring programs and predictive models are required to provide an ongoing assessment of the state of the aquatic ecosystems? These programs must ensure all stakeholders have the opportunity for input.

Non-Scientific Questions

15) How can the Study results be communicated most effectively?

16) What form of interjurisdictional body can be established, ensuring stakeholder participation for the ongoing protection and use of the river basins?

APPENDIX C

NORTHERN RIVER BASINS STUDY

TERMS OF REFERENCE

Project 5201-C1: Assessment of Population, Community and Ecosystem-level Approaches and Metrics to Assess and Monitor Aquatic Ecosystem Health

I. Introduction

One of the main objectives of the Northern River Basins Study is to determine the cumulative effects of industrial development on the Peace, Athabasca and Slave river systems. As part of this exercise, methods and approaches must be identified to assess and monitor aquatic ecosystem health.

II. Requirements

1) Review the concept of ecosystem health and cumulative effects assessment and the underlying theoretical framework and practical objectives of these approaches.

2) Critically review the literature on existing population, community and ecosystem-level approaches and associated metrics being used for the assessment of aquatic ecosystem health and cumulative effects. Outline the shortcomings and advantages of each.

3) Identify the types of data and information required to adequately assess and monitor aquatic ecosystem health and cumulative effects.

4) Identify currently available analytical packages which facilitate the assessment of ecosystem health.

5) Assess the applicability of these approaches and metrics to the Northern River Basins Study and recommend approaches that could potentially be employed to assess and monitor aquatic ecosystem health and cumulative effects within the Northern River Basins.

III. Reporting Requirements

1. Submit five copies of a draft report outlining the information specified in section II to the component coordinator by January 1st, 1994.

2. Three weeks after the receipt of review comments on the draft report, submit five cerlox bound copies and two unbound, camera-ready copies of the final report to the certification officer. At the same time submit and electronic copy, in Word Perfect 5.1 format and on 5¼ or 3½ inch floppy disk, of the final report to the certification officer. An electronic copy (Dbase IV format on floppy disk) of data used to develop figures, tables and appendices in the final report is also to be submitted to the component coordinator. The final report is to include an executive summary, table of contents, list of tables, list of figures and an appendix which includes the Terms of Reference for this project.

IV. Project Administration

The component coordinator for this project will be:

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