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A Framework to Improve the Effectiveness of Aquatic Environmental Impact Assessment

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INTRODUCTION

This study, which concerns environmental impact assessment methodology and technical aspects of aquatic contamination studies, was funded by a Canada Department of Supply and Services research grant, sponsored by Environment Canada, Department of Indian Affairs and Northern Development, the Federal Environmental Assessment Review Office, the Northwest Territories Water Board and Echo Bay Mines Ltd. The study was initiated to aid practitioners of aquatic impact assessment by transferring current impact assessment concepts to the applied arena of aquatic contamination studies in Canada.

Several terms appear throughout the report. Their definitions in the context of this study are given below.

- Environmental Impact Assessment (EIA) encompasses both the use of baseline information to predict environmental consequences and follow-up monitoring to test predictions. It refers to overall environmental assessment and planning which may be required under a variety of jurisdictions rather than a specific formalized EIA process.

- Framework, conceptual framework and conceptual model are used interchangeably to refer to the framework developed by Hakanson which identifies the basic components of an aquatic contamination problem and describes their relationship to one another. The framework provides a system for rationally analyzing a contaminant problem, designing an assessment study and gathering relevant data.

- Risk index is an expression of ecological risk associated with a given contamination problem which permits ranking of contamination problems between lakes and ranking of contaminants within lakes.

- Diagnostic/prognostic model for metal contamination in fish is a quantitative (empirical) expression of an ecological effect (mercury in fish tissue) resulting from a known contaminant dose (mercury in surficial sediments).

THE PROBLEM

The diverse expectations of EIA's main participants (government administrators, project proponents, scientists, consultants) are seldom met by traditional assessment methods practiced in Canada; however, time and budget constraints discourage experimentation with new techniques.

Aquatic pollution control is based largely on arbitrary discharge criteria derived from laboratory toxicity bioassays. Studies are frequently oriented towards compliance monitoring; large volumes of data are generated, very little of which are analyzed and applied to quantitative prediction and assessment of contaminant effects and related resource management.

Improvements to aquatic contaminant impact assessment studies are required to;

- develop consensus among study participants on assessment priorities and study procedures,
- systematically build up information on the behaviour of contaminants in receiving waters,
- generate information which is relevant to resource managers,
- permit quantitative impact predictions which can be tested, and
- generate time and space comparable data.

STUDY OUJECTIVES

The purpose of this study was to illustrate the benefits of the conceptual framework **for aquatic contaminant impact assessment for improving impact assessment methods, resource management capabilities and the scientific quality of aquatic contamination studies.**

A conceptual framework being developed by Dr. **Lars Hakanson of the National Swedish Environmental Protection Board was described and used** as a template for assessing a typical Canadian contaminant study (Lupin Mine, N.W.T.) and identifying areas for improvements. Data from 14 Canadian contaminant studies were reviewed and assessed for applicability to the development of contamination factors, ecological risk indices and diagnostic/prognostic models for metal accumulation in fish tissue. Areas for improvement in study design, sampling methodology and data analysis to aid resource management and impact prediction were identified.

THE FRAMEWORK

The conceptual framework for assessing the effects of aquatic contamination provides a systematic outline for tracing contaminants from dose to response and accounting for the mediating influences of environmental factors. The framework is based on geoecological principles which provide a sound rationale for study design (e.g. sample site selection, sample numbers, standards for collection/analysis) to generate time and space comparable data.

The components of Hakanson's framework are expressed by the following notation:

$$E = f(D, T, W_i) + R$$

where:

E = a parameter expressing an ecological effect.

The framework approach advocates selection of a limited number of information-rich effect parameters. Criteria for selection include:

- Generous representation of the potential effect field by parameters which integrate a number of sub-effects (e.g. a tertiary consumer).
- Representation of, or direct linkage to, valued ecosystem components.

- Representation of sensitive or weak links in the contaminant pathway.

D = a contaminant dose parameter.

Depending on the contaminant pathway and the effect parameters of concern, the dose may be expressed in various forms. It may be defined by a simple mass balance equation expressing input flow volumes and concentration of contaminants, or it may integrate information concerning its expression in the receiving environment (e.g. concentrations in sediments, aquatic vegetation, etc.)

T = a factor expressing toxicity.

The toxicity factor is a measurement of contaminant toxicity as it is expressed in natural waters (e.g. based on natural abundance in the receiviny water body).

W_i = The ith factor or parameter expressing receiving water sensitivity.

Sensitivity parameters are those factors in natural receiving waters (e.g. pH, O_2, alkalinity, salinity, bioproductivity, water exchange) which influence the distribution of contaminants in receiviny environments and the ecological response to contaminants.

R = a residual term (the unaccountable remainder; a measure of the model's effectiveness).

The residual term (R) represents natural variability and environmental factors, unaccounted for in our study design, which yive rise to a difference between predicted and observed responses of aquatic systems to a contaminant dose. The objective of the framework is to express quantitatively (by theoretical, empirical, intuitive or mathematical models) normative E-values from a limited number of readily measurable and representative integrating variables and to minimize the residual term. The R term provides us with a measure of our success in accounting for the major operative mechanisms and the natural variability of the studied ecosystem.

The relationship of contaminant dose to environmental response is also influenced by space and time. The effect terms may be transformed to indices by integratiny spatial aspects of the effect. Using the conceptual framework notation;

$$E'' = \int^{A} E' \quad \text{or} \quad E' = A \cdot E'' + R$$

E'' stands for the potential ecological risk accounting for "biological contact area". The greater the area affected by aquatic contamination, the greater the potential ecological risk.

The biological contact area (A) provides spatial dimensions to the dose, such as the physical area of influence of contaminant dose and the biologically available fraction of the contaminant dose. An understanding of the physical processes (e.g. those giving rise to erosion, transportation and accumulation zones in lake sediments) is required to define the physical area of contaminant influence. Definition of the biologically available dose may require a simple standardized fractionation analysis yielding, for example, exchangeable, organic and inert fractions of contaminant dose.

Incorporation of temporal aspects of the contaminant dose or "biological contact time" (P) acknowledges the assumption that the ecological effect varies directly with the duration of the contamination (under otherwise comparable conditions). Integration over time yields;

$$E''' = \int^{P}\!\!\int^{A} E' \quad \text{or} \quad E''' = P \cdot A \cdot E' \text{ t } R$$

where E''' stands for the potential ecological risk accounting for time and area.

The "additive effects" of different toxic substances polluting the same receiving water are incorporated in an overall index as follows;

$$PER = \sum E''' + R \quad \text{or} \quad PER = \sum P \cdot A \cdot E' \text{ t } R$$

where PER is the potential ecological risk index for a given set of contaminants in a given receiving water. Effects may not be strictly additive.

Discrepancies between observed and expected effects (large R) may reflect synergism or antagonism and suggest areas for further investigation.

The conceptual framework is illustrated schematically in Figure 3 with framework examples of parameters for each component. Various elements of the framework can be selectively analyzed to meet specific assessment objectives. Analytical tools developed from the framework to date include contamination factors, risk indices and diagnostic/prognostic models.

A CANADIAN CASE STUDY

The Lupin Mine case study is a typical Canadian aquatic contaminant study required to obtain a water licence. The Lupin study was critically reviewed using the framework as a template to identify opportunities for increasing the value of information obtained from this and other such studies.

Pertinent features of the Lupin study included the following:

- Work addressing the dose and response components of the study was carried out as waste management/treatment design and aquatic contaminant studies respectively.
- The waste management studies concerned mine tailings treatment design to achieve pre-specified effluent criteria.
- The aquatic studies involved characterization of the fisheries resource in the receiving water bodies and collection of baseline data for future effects monitoring.
- Environmental impact prediction was not an objective of the study but was ultimately required to negotiate a higher effluent concentration criteria for zinc and assess arsenic toxicity.

FRAMEWORK EVALUATION OF THE CASE STUDY

The framework analysis of the Lupin studies identified areas for improvement as follows:

- Initially the Lupin study objectives avoided the issue of impact prediction and were oriented towards regulatory monitoring. When the need to address practical management problems arose, impact prediction was inevitable and new information requirements became evident.

  The framework orients the study design towards practical objectives at the outset. It presents a series of assessment milestones with examples of corresponding study products (e.g. impact prediction models, effects monitoring, contamination factors and ecological risk indices). Thus, the framework approach encourages the study design to address these potential practical applications:

- In the Lupin study, belated information on the nature of the dose shifted the aquatic study objectives from an effects monitoring focus to an impact prediction focus. As a result, some of the upstream sample sites were rendered extraneous; some important information had not been obtained (e.g. fractionation of the contaminant dose).

  The fundamental principal of the framework is the linkage of dose with response. The full integration of these components is necessary to isolate critical study parameters and minimize extraneous data collection.

- The Lupin studies were oriented towards baseline data collection, with little impetus to explore relationships between effect and sensitivity parameters or to compare these relationships with other available data bases. Consequently a large amount of data was generated, very little of which was applied to analyzing contaminant impacts.

  The framework directs study towards the development of prognostic tools by exploring empirical relationships between dose, recipient sensitivity and effects (Section 7). Developing hypotheses for these relationships provides a rational basis for selecting sensitivity parameters and exploring the data bases for these relationships. This ongoing evaluation and application of the data base provides a check on extraneous data collection.

- The effect terms initially chosen for the Lupin studies were metal concentrations in water, sediments and fish tissue and invertebrate community structure. When it was known that the dose would occur at high flow for a short period, the framework was used to evaluate the chosen effect terms. Due to the transportational character of the sediments, the mobility of the fish, and the natural temporal variability of invertebrate communities, it became clear that the only term that was likely to integrate a measurable effect during the monitoring period was water. This was considered an inadequate representation of the potential effect field. Therefore, metal accumulation in clams and snails was added as an effect term which would potentially integrate an effect over a short time period and provide a link to the effect term of primary concern, the fish. There was equal justification for limiting sediment analysis to accumulation areas, such as an accumulation basin on Contwoyto Lake near the outlet of Sun Bay. The results of this refinement were more information gained for effort expended and measurement of effect at a "weaker" or more sensitive link in the contaminant path which, if it persisted, might ultimately be related to metal contamination in fish.

- The framework provides some tools for expressing effects (e.g. contamination factors) which, if widely adopted, could be used to compare the behaviour of contaminants in different systems. Accumulation of such comparative information may ultimately be very valuable to resource managers.

- The Lupin study program did not explicitly address the toxicity, contact area, contact time and additive effects components of the framework, although systematic consideration of these factors helped in data interpretation. By using a framework which incorporates these components the investigator is encouraged not only to address them in his study design but to seek relevant information from the dose component of study.

THE ECOLOGICAL RISK INDEX

The ecological risk index is a diagnostic tool, provided by the framework, which ranks lakes according to the degree of ecological risk associated with their contamination and ranks the contaminants within each lake. This information allows resource managers to focus study and/or mitigation initiatives on the lakes and contaminants posing the highest risk.

The risk index is derived from the following parameters:

- The dose - expressed as a contamination. factor (C_f^i)-derived from the contaminant concentration (C^i_{0-1}), (based on five surficial sediment samples from accumulation zones) divided by the pre-industrial contaminant concentration (C^i_n), (defined as the mean plus one standard deviation for elements in uncontaminated sediments from 50 European and American lakes); i.e.,

$$Cfi = \frac{Ci_{0-1}}{Ci_n}$$

- The receiving water sensitivity - as a bioproduction index (BPI) determined from nitrogen content and ignition loss in sediments.

- The toxicity of the contaminant ($S_t{}^i$) - based on its natural abundance in nature, and its "fingerprint" in sediments (the quotient between pre-industrial concentrations in water and sediment). A toxic response factor (Tri) is derived which accounts for the sediment toxic factor (Sti) and the effect of BPI on the toxicity of the contaminant.

Based on these parameters, a risk factor for each contaminant is defined as the product of the toxic response factor and the contamination factor; i.e.,

$$Eri = Tri \cdot C_f{}^i$$

The potential ecological risk index (RI) is then defined as the sum of the risk factors for eight individual contaminants; i.e.,

$$RI = \sum_{i=1}^{8} Eri = \sum_{i=1}^{8} Tr^i \; . \; C_f{}^i$$

RISK INDICES FOR CANADIAN LAKES

Risk indices were developed from data for 14 Canadian lakes. The data were incomplete for use in the risk indices; therefore, some assumptions were made and the risk indices were calculated for illustrative purposes. The risk indices appeared to rank the lakes appropriately according to available information for the lakes. The risk index analysis yielded the following observations:

- There is good potential for applying data from across the country to comparative observations on the occurrence of contaminants in various aquatic systems. By relating these observations to their respective contaminant discharge scenarios, strong empirical evidence can be compiled for guiding future contaminants management.

- Sampling design and protocol in Canada is inconsistent and limits use of data for comparative analysis. Recommendations for standard methods include sampling sediments in accumulation zones, analyzing the surficial (0-1 cm) layer and including consistent measurements of recipient sensitivity (e.g. BPI as nitrogen content and ignition loss in sediments).

- Regional values for pre-industrial 'contaminant' concentrations could be developed to give higher resolution to contaminant factor analysis.

The analysis also highlighted areas for further investigation and development:

- Risk indices should be developed for contaminant groups of specific concern to Canada. Separate indices could be developed for metals and organic pollutants.

- The risk index should be developed to incorporate direct measures of ecological risk (i.e. biological effect terms).

- Risk indices should be developed for other physical regimes (fluvial, estuarine, marine, etc.).

DIAGNOSTIC AND PROGNOSTIC MODELS FOR AQUATIC CONTAMINATION

The derivation, from the framework, of a model for mercury contamination in fish was described and the model was tested using available Canadian data. The model, or empirical formula, includes the following components:

- The dose term - weighted mean mercury content of surface sediments (0-1 cm) in ng/g dry substance (Hg_{50}).

- The sensitivity parameters - pH and bioproduction index (BPI) as determined from sediments or total phosphorus in water.

- The effect term - the content of methyl-mercury in 1 kg pike muscle tissue as ng/kg wet weight [F(Hg)].

The derived formula is:

$$F(Hg) = \frac{4.8 \times \log(1 + Hg_{50}/200)}{(pH - 2) \times \log(BPI)}$$

Available Canadian data suitable for testing the formula was limited to six lakes. Data for mercury content in sediments, pH and BPI were analyzed to derive mercury levels in pike tissue which were then compared to measured values. The predicted values corresponded well with measured values for four of the six lakes. For the remaining two lakes, measured values were low compared to predicted values, suggesting possible lead or zinc antagonism. High concentrations for those elements in the lakes corroborated that hypothesis.

The problems of inconsistent and incomplete data for Canadian lakes were discussed. Recommendations for improving the data base were given.

Examples were given to illustrate the application of the formula for assessing the implication of sewage treatment to metal contamination in fish, or in assessing environmental effects and mitigation requirements related to acidification of lakes.

APPLICATION OF THE FRAMEWORK APPROACH

The study demonstrated that there are many benefits to adopting the framework approach, particularly in applied studies of aquatic contaminants related to impact assessment and resource management. Use of the framework approach would best be implemented by a government agency responsible for waste permit approvals and environmental protection. Such an agency would be able to standardize sampling methods to generate comparable data and stipulate routine measurements of sensitivity parameters.

Using the framework and its associated tools (contamination factors, risk indices, models), agencies could develop information with a broader application than compliance monitoring. They could systematically build a data base directly applicable to resource management. It is strongly recommended that an existing resource management agency or a specially constituted workshop of agency representatives test the framework approach on the design of an aquatic contamination study.

RECOMMENDATIONS FOR FURTHER STUDY

- The framework approach encourages refinement and further development of all linkages. In particular, work is required on the integration of contact area and contact time. Information is required on physical dynamics and the fractionation characteristics of contaminants in the receiving environment.

- Development of prognostic and diagnostic models for contaminants is a major area for further work. Available data should be assembled, normalized and augmented by standardized sampling methods.

- The use of new effect terms (e.g. contaminant concentrations in aquatic plants and clams, production-respiration ratios, etc.) should be explored both for biomonitoring application and tracing contaminant pathways.

- Future data gathering programs for aquatic contaminant studies should include;

    - contaminant dose data in conjunction with effect term data,
    - contaminant fractionation data,
    - contact area data, and
    - sensitivity parameters data.

- Implementation of standard data collection methods and protocol is required to generate comparable data.

- Regional data should be gathered for pre-industrial levels of contaminants in Canada.

- A major area for further work lies in the application of the framework approach to other physical regimes (rivers, marine environment, etc.).

CONCLUSION

Basic concepts of ecological impact assessment have been widely discussed in the technical literature; however, routine integration of these concepts into applied investigations has been slow to occur. The framework identifies and organizes key information components with an impact assessment problem. As such, it may be used as a template to help investigators systematically address and rationalize these important components in their study design. It also provides a common frame of reference for EIA participants to communicate and coordinate their study objectives. Although the framework has a simple structure, it can accommodate sophisticated reasoning. It invites further development and refinement. The framework can be readily implemented to improve impact assessment capability and strengthen the scientific basis for applied aquatic contaminant investigations.

1.0 INTRODUCTION

1.0 INTRODUCTION

The impetus for this study arose from our desire, as practitioners of Environmental Impact Assessment (EIA)* pertaining to aquatic systems, to improve the tools of our trade. As consultants specializing in water resource management, our aims are to facilitate responsible resource development while applying our specialized knowledge in aquatic sciences to the effective stewardship of aquatic environments. A key to fulfilling these tasks is aidiny communication and developing agreement amongst project proponents, government administrators, technical experts and the public regarding appropriate assessment methods and management actions.

In pursuing these objectives we have met with constraints commonly encountered by practitioners of aquatic environmental assessment, primarily, the rudimentary state of impact prediction and the complex and, at times, arbitrary administrative frameworks for water management. Despite the common shortcomings of the aquatic impact assessment process, many valuable insights and techniques have arisen from isolated studies. Those advancements now need to be introduced, worked with and developed by the aquatic impact assessment community at larye. Though our understandiny of aquatic systems responses to perturbations is far from complete, the vital job of aquatic resource management must still be carried out. We require practical tools for aquatic pollution control in the short term and at the same time we need to systematically build our scientific understanding of aquatic impacts.

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* EIA as used in this report emcompasses both the use of baseline information to predict environmental consequences and follow-up monitoring to test predictions. It refers to overall environmental assessment and planning which may be required under a variety of jurisdictions rather than a specific formalized EIA procedure.

The purpose of this work is to illustrate the use of a framework for aquatic contaminant impact assessment currently being developed in Sweden by Dr. Lars Hakanson of the University of Uppsala and of the National Swedish Environmental Protection Board. This framework is applied to a current Canadian aquatic contaminant case study to explore its potential for improving our impact assessment and prediction capabilities and for building up our understanding of aquatic impacts.

The term framework as used in this report refers to the overall conceptual model, developed by Hakanson, which identifies the major elements of an aquatic contaminant system and describes their relationship to one another. Briefly, the framework expresses the ecological effects (E) of aquatic contamination as a function of the contaminant dose (D), the contaminant toxicity (T) (as expressed in natural waters) and the sensitivity of the recipient aquatic system to the contaminant (W_i). A residual term (R) accounts for the practical impossibility of developing a model capable of giving a complete explanation of ecological cause-effect relationships. Time, space and additive effects are also incorporated in the framework. This framework is not a simulation model but a system for rationally analyzing a contaminant problem, designing an assessment study and gathering relevant data.

The framework supplies the logic to identify and quantify various terms and relationships of the contaminant system and to carry out different orders of analysis for specific impact assessment applications. Two types of analysis which will be discussed in this report are an ecological risk index (Section 6) and a diagnostic/ prognostic model (Section 7) for metal contamination in fish tissue.

The risk index is a diagnostic tool which provides a quantitative value for the potential ecological risk associated with a given aquatic contamination situation. It permits rankiny of contamination problems between aquatic ecosystems and ranking amongst the contaminants of concern.

The diagnostic/prognostic model for metal contamination in fish is a quantitative (empirical) expression of specific components of the framework allowing quantitative prediction of an ecological effect (e.g. mercury concentrations in pike tissue) resultiny from a known contaminant dose (e.g. measured as mercury concentrations in the top 1 cm lake sediments).

A Canadian case study (Section 5) is reviewed in the context of both the conceptual framework and its accompanying quantitative tools to illustrate how the framework approach can enhance impact study design, data analysis and the scientific basis for aquatic impact assessment. The case study presented was chosen because of the authors' familiarity with the study and because we believe that it is a typical example of contaminant discharge assessment studies in Canada. The procedures for technical studies in support of licenciny are constantly evolving in response to new information and objectives. It is hoped that this project will contribute to the transfer of technology and the evolution of these procedures.

We emphasize that the conceptual framework as illustrated is not intended as a rigid structure providing the only approach to aquatic contaminant impact assessment. Rather it is like a skeleton which can be built upon and modified as our understanding of contaminant systems grows. Furthermore, the framework is not intended to provide a blueprint for ecological research. It is intended to generate practical tools for aquatic impact manayement from a <u>limited</u> number of readily and inexpensively measured and representative integrating variables. It does, however, incorporate a strong scientific rationale and as such can significantly enhance the scientific validity of impact assessment studies.

2.0 BACKGROUND

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2.1 THE STATUS OF ENVIRONMENTAL IMPACT ASSESSMENT

Recent reviews of tne status of environmental impact assessment (EIA) in Canada agree that major changes are needed to improve the effectiveness of the process (Efford 1976, Rosenberg and Resh 1981, Beanlands and Duinker 1983). Currently EIA is guided by procedural frameworks established by government policies and legislation; however, there are no commonly accepted scientific/technical standards for the content of assessment studies. Similarly there are no commonly employed procedures to create fruitful collaboration among the main EIA participants to design and execute an optimal EIA study (Beanlands and Duinker 1983).

This lack of commonly accepted standards for the content of impact assessments has given rise to diverse expectations amony the main participants in EIA about the purpose and function of the process. The major participants identified by Beanlands and Duinker (1983) and their perspectives on EIA include:

- <u>The Government Administrators</u> - who tend to view EIA as fulfillment of procedures set by policy and legislation (Beanlands and Duinker 1983).

  In addition, the administrators are responsible for usiny the information generated by EIA to make resource management decisions (project yo or no yo; if yo, under what mitigatory or compensatory conditions?). Faced with complex issues requiring multi-disciplinary investigations they may or may not have the technical background to specify the scientific inquiries they require to aid assist resource rnanayement decision-making.

Administrators may enlist scientists to prepare terms of reference for EIA or to review the study results but, without the context of social and resource management concerns or the practical constraints of study (time and money), this input can be misleading and at times disruptive (Beanlands and Duinker 1983).

- The Project Proponents - view EIA as a necessary precursor to project approvals and in some cases as a means of enhancing public relations (Beanlands and Duinker 1983).

  EIA's may be very costly and may lead to imposition of safety factors in project design which could significantly affect project feasibility (de Broissia 1984).

  Proponents frequently hire consultants to carry out the EIA. The proponents may not be fully aware of the contents or rationale for EIA or understdnd how their active participation in the process could save them money or enhance their project planning process.

- The Research Scientists - often feel that the political and time constraints acting upon EIA studies preclude the conduct of acceptable science (Beanlands and Duinker 1983). They may be involved in EIA for isolated tasks (e.g. report review, development of terms of reference) but seldom are their skills used to best advantdge. EIA will never (and probably should never) enjoy the relatively unrestrained circumstances of pure research projects; however, the effective application of scientific knowledye and methods to the applied field of EIA is essential if the process is to become a more powerful and useful environmental manayement tool.

- The Consultants - are frequently the practictioners of EIA on behalf of project proponents. They are in the position of having to (1) fulfill EIA procedural requirements; (2) address political issues; (3) minimize the costs of the study to the client; and (4) meet the standards of scientific and technical reviewers who may or may not have been involved in earlier phases of the study (Beanlands and Duinker 1983). The consultant, attempting to reconcile the diverse expectations of the preceding groups, is more apt to resort to established precedents for EIA methods than risk study time and budget in experimenting with new approaches.

Traditional EIA methods, such as the "busy taxonomist" and "information broker" approaches described by Valeila-Ward (1978) have relied primarily on descriptions (in some instances highly comprehensive descriptions) of pre-project environments followed by judgements and deduction of probable project effects. There have rarely been post-development studies to determine whether impact predictions, even qualitative ones, were on track (Clark 1983, Bisset 1980, 1982). A small percentage of EIA's have explored more definitive techniques with success (Holling 1978, Beanlands and Ouinker 1983, Valeila-Ward 1978) but on the whole EIA and our abilities to predict environmental impacts have advanced very little.

Beanlands and Duinker (1983) described EIA in Canada as being at a cross-roads...

> "either we improve scientific rigour of the studies which support the entire process, or we run the risk of seeing the concept degenerate into an exercise in public relations and government lobbying."

Recommendations for improvements to EIA include the following:

- Use of a procedure (e.g. modelling workshops) whereby diverse interest groups can communicate effectively and reach a consensus on assessment priorities and study procedures.

- Clear definition of the study objectives, valued ecosystem components and effect parameters of concern.

- Definition of a temporal, spatial and statistical context for prediction and measurement of impacts.

- Development of a study strategy focussing on clearly rationalized linkages between the project and the effect parameters of concern.

- Formulation of explicit, quantitative impact predictions.

- Prediction testing (e.g. by experimentation and project monitoring).

2.2 AQUATIC CONTAMINATION IMPACT STUDIES

The essential objective of aquatic contamination impact studies is to determine the effect of contaminant discharges on the receiving environment. Elements of the 'receiving environment' of importance to resource managers may include aquatic biota (especially fish) and man, through his use of aquatic resources. A common approach to aquatic contaminant studies is to examine various physical, chemical and biotic components of the receiving waters to characterize their value and sensitivity to contaminant discharge and to generate baseline data for subsequent effects monitoring. The contaminant discharge or dose is described by flow and composition and attempts are made to predict its ultimate disposition in the receiving waters. These predictions may range from qualitative to quantitative, based, for example, on the results of various types of simulation modelling. Frequently, quantitative predictions on the fate of contaminants in the physical environment (e.g. concentration, dispersion area) are used to generate qualitative predictions of effects on aquatic biota and man systems.

Much of the effort in aquatic contamination studies is geared towards ensuring that contaminant discharges meet established standards for chemical concentrations either at the end of the discharge pipe or in the receiving waters. The standards strive to protect aquatic life based on toxic threshold concentrations determined by laboratory bioassay studies. Since the standards generally must be met by contaminant discharges and should, by definition, mitigate against acute toxic impact in the receiving environment, contaminant impact assessments are frequently relegated to predictions of subtler and more complex chronic and cumulative effects.

There are two major problems with this traditional approach. In the first place, while our standards for contaminant control are based primarily on laboratory toxicological tests, we lack information on the ways in which environmental factors alter the expression of contaminant effects in natural receiving waters (i.e. observations in laboratory toxicology tests may not be transferable to the field). Thus to date we do not have a strongly developed rationale for setting of effluent control standards.

Secondly, we lack the information and tools with which to explicitly state and test predictions of sub-acute impacts or express these impacts in terms which are meaningful to resource managers and decision makers. Resource managers may be faced with decisions such as whether to require costly waste treatment of an industry to meet arbitrary discharge standards or to risk unknown effects on the receiving environment by relaxing discharge standards. Qualitative assessments, such as, "may be some adverse effect on fisheries production or quality" do not aid the decision-makiny process nor do they provide any yardsticks by which to test and refine the accuracy of impact predictions. Quantitative predictions, while difficult with our present understandiny of aquatic ecosystems, are required to introduce accountability to the impact assessment and resource management process.

Quantitative assessment and prediction of aquatic impacts are plagued by data problems related to the patchy distribution of organisms and natural variability of conditions over time (de Broissia 1984). To build on our contaminant impact assessment experience we require the development of sampling protocols which will generate time and space comparable data. This in turn imposes the requirement for a sound understanding and integration of physical and biotic components of aquatic ecosystems. Frequently aquatic impact assessments fail to develop adequately the physical rationale for sampling programs.

With these deficiencies, we are still largely in the realm of free speculation when faced with predicting the impacts of contaminant discharges or prescribing appropriate aquatic pollution control standards. While isolated studies have used methods which attempt to address these deficiencies, there is now a need, amongst the community of aquatic EIA practitioners as a whole, to define fundamental assessment study requirements and to systematically build upon our collective impact assessment experience. The framework for aquatic impact assessment is presented as a possible foundation on which to design contaminant impact assessment studies and to build our understanding and predictive capabilities related to aquatic contaminant impacts.

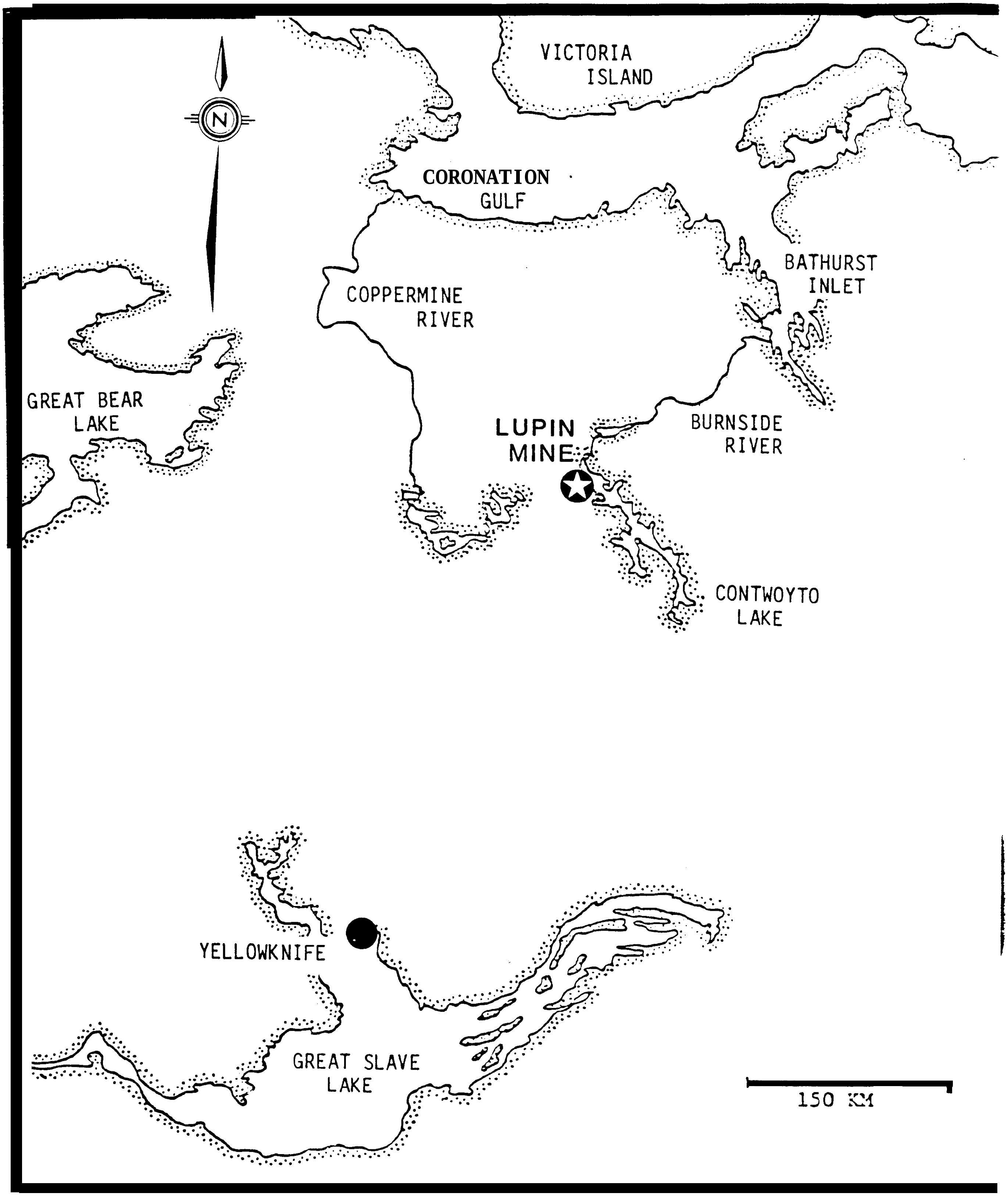
2.3 A CANADIAN CASE STUDY

The use of the conceptual framework for designing procedures and interpreting the results of aquatic contaminant studies is illustrated by applying the framework to aquatic studies at a gold mine currently operating in the central barrenlands of the Northwest Territories. Lupin Mine, operated by Echo Bay Mines Ltd., is located on the shore of Contwoyto Lake near the northwestern end of the lake

(Figure 1). Approximately thirteen hundred and fifty tonnes of ore are processed daily. The gold is extracted by a cyanidation process and mill tailings are discharged to a large tailings impoundment basin 6 km south of the mine. Based on the current mill operating rate, the tailings impoundment has a minimum two year holding capacity. The impoundment consists of two ponds in series. Solids are allowed to settle in the upper pond and the decant fluid is treated with an iron salt to enhance precipitation and settlement of metals in the lower pond. Cyanide is reduced by natural aeration (and probably by photo-oxidation). Tailings water is decanted from the lower pond to a receiving stream basin (Seep Creek) west of the tailings impoundment by means of five siphons with a total discharge capacity of approximately 2 m^3/s. Decanting operations are intermittant and occur over a period of one month on an annual basis. The contaminants of interest in the tailings discharge are cyanide and the metals arsenic, zinc, copper, nickel, lead and iron.

The immediate recipient of the decant discharge is a small shallow lake, called Dam 1A Lake, which in turn discharges to the 6 km long Seep Creek and ultimately, via the small Unnamed Lake, to Sun Bay on Contwoyto Lake (Figure 2). Seep Creek, just above Unnamed Lake, has an average discharge of 0.26 m^3/s during open water and a measured range in flows from 0.02 m^3/s in the fall to 4 m^3/s during spring runoff. Several small tributaries join Seep Creek downstredm of the tailings pond and a major stream, Concession Creek, discharges to Unnamed Lake from the southwest. Average annual flows in Concession Creek are estimated at 2.5 m^3/s. Sun Bay is approximately 4 km2 and divided by a narrow constriction into a shallow (1-4 m deep) inner basin - Inner Sun Bay, and a deeper (20 m deep) outer basin-Outer Sun Bay (Figure 2).

Conditions governing the water supply and wastewater disposal for Lupin Mine are specified in a Northwest Territories Water Board



LUPIN MINE
LOCATION MAP

FIGURE 1

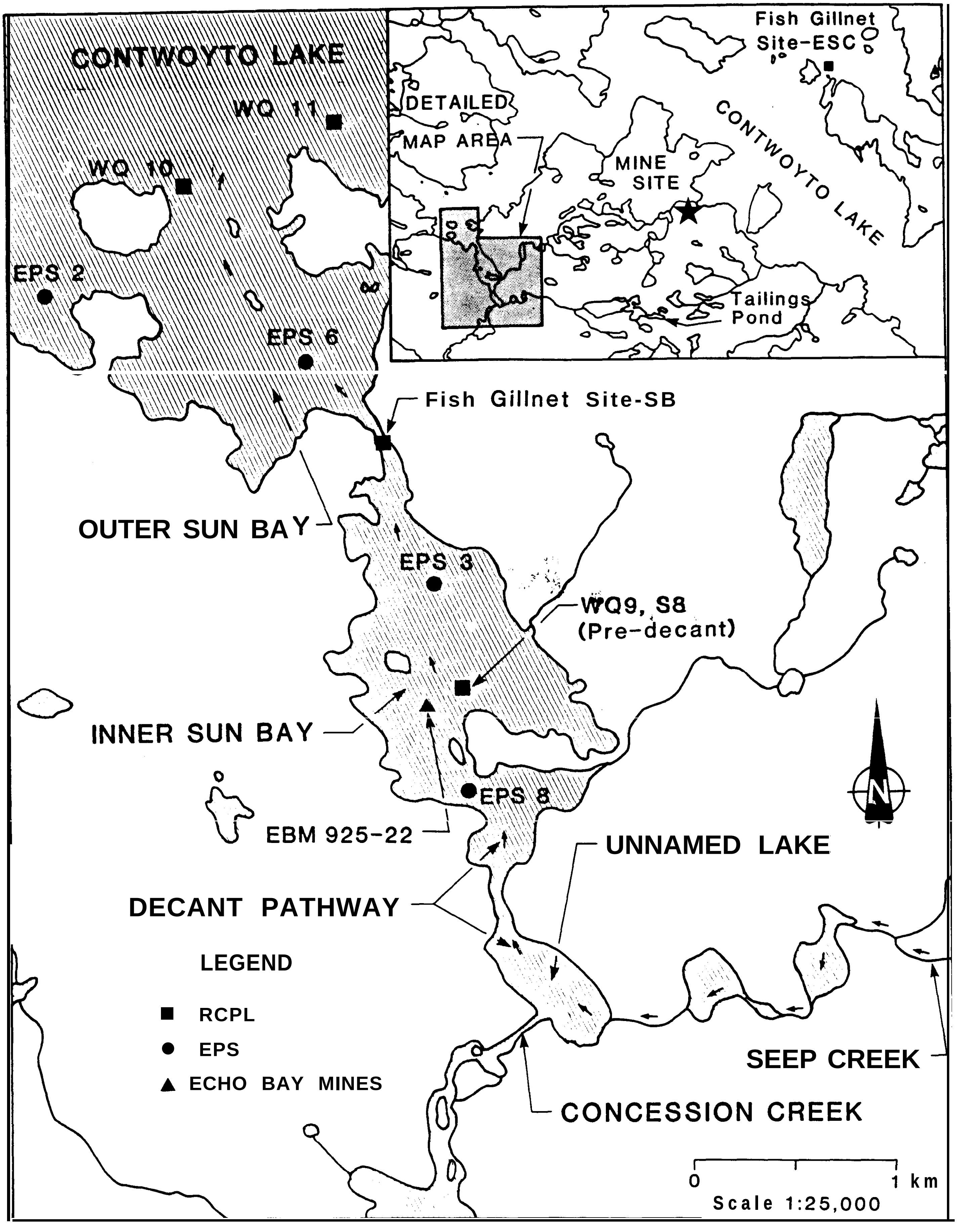


Figure 2: Detailed Study Location Map With Sampling Sites.

Licence, enforced by the Water Resources Division of Indian Affairs and Northern Affairs Canada. The pertinent conditions, are as follows:

- the requirement to determine the chemical, physical and biological properties of the aquatic environment potentially affected by the project operations,; and

- the requirement to characterize the liquid waste in the tailings contaminant area and to assess alternative methods for waste treatment to meet effluent quality standards specified by the Northwest Territories Water Board (and written in the Water Licence).

The studies addressing these conditions were carried out by consultants for Echo Bay Mines Ltd. (Reid Crowther 1985A,B, R.L. & L. Environmental Services Ltd. 1985). The studies were designed by the consultants in consultation with the Technical Advisory Committee of the Northwest; Territories Water Board, whose members represent Environment Canada, Department of Indian Affairs and Northern Development, Department of Fisheries and Oceans, Health and Welfare Canada, Government of the Northwest Territories - Department of Renewable Resources: Northwest Territories Association of Municipalities and the Northwest Territories Chamber of Mines.

The aquatic studies for Lupin Mine commenced in 1980 and comprised a baseline inventory with considerable emphasis given to development of a statistically defined database for use in future project monitoring. Impact assessment was not an explicit objective of the studies although the parameters measured implicitly reflected environmental concerns (i.e. water quality, benthic productivity, accumulation of metals in sediments and fish).

The waste management studies were orientated towards design of waste treatment processes to meet the effluent standards stated in the

Water Licence and based on federal metal mine standards. The effluent standards were stipulated before the aquatic studies commenced.

Although the aquatic and waste management studies were carried out under separate objectives the findings of the two studies were ultimately combined to produce an impact analysis prior to the hearings for Water Licence renewal. This analysis was used to rationalize revision of the effluent standards and the proposed design and operation of tailings treatment facilities. Issues concerning the established effluent standards included the appropriateness of the standards for zinc and total arsenic.

The studies conducted for Lupin Mine were typical of, though possibly somewhat more comprehensive than, the majority of mine water licence studies conducted at that time. In hindsight, the design and execution of these studies suffered from a number of the shortcomings mentioned in previous sections:

- Varied expectations amongst the participants regarding the objectives and content of the study. The methodology for the study evolved in an ad hoc manner, in response to individual concerns expressed by members of the Techical Advisory Committee and to logistics, technical and budgetary constraints.

- Lack of a framework within which to resolve expectations, reach a concensus on appropriate study methods and focus efforts on the most significant environmental issues. Different opinions about the appropriate scope of study, held by the proponents and individual members of the Technical Advisory Committee, persisted throughout the study.

- Failure to orient the study explicitly towards impact prediction as a basis for hypothesis testing (monitoring), formulation of appropriate effluent standards and waste treatment design.

- Lack of a systematic framework to assign appropriate dimensions to the study and hence generate time and space comparable data.

While aquatic studies such as those for Lupin have been considered adequate in the past, we feel that there are opportunities to significantly enhance the efficiency and practicality of aquatic contaminant studies and increase the value of information gained therefrom. In this report, we use the Lupin Mine case study to demonstrate how the application of a systematic framework may accomplish such improvements.

3.0 STUDY OBJECTIVES

3.0 STUDY OBJECTIVES

The purpose of this study was to introduce a conceptual framework for the design and execution of aquatic contaminant studies with emphasis on impact assessment requirements and scientific validity. To enhance the applicability of studies to EIA, a conceptual framework must:

- Allow clear definition of study objectives and their relevance to aquatic resource management.
- Assist study participants of diverse backgrounds to communicate and develop a concensus on study approach and procedures.
- Assist selection of appropriate "effect terms" including valued ecosystem components.
- Assign explicit dimensions to the aquatic contamination problem.
- Focus study efforts to maximize information gained relative to study costs.

To enhance the scientific validity of aquatic contaminant impact studies, a framework must:

- Facilitate a clear definition of cause-effect pathways and a clear rationalization for choice of parameters measured (i.e. leave a clear trail of scientific rationale which can be built upon).
- Facilitate quantification of impacts.
- Facilitate expression of impacts as testable hypotheses (predictions).

Such a framework, if widely recognized and applied, would permit us to systematically build our understanding of aquatic contaminant processes and to develop powerful tools in aquatic pollution control.

This study examined a conceptual framework for aquatic contaminant impact assessment, developed by Dr. Lars Hakanson. The framework provides the foundation from which the basic information requirements of aquatic contaminant studies can be assessed in an orderly and logical fashion. Using the notation system supplied by this framework, logical analytical themes can be developed with varying degrees of complexity and power in their applications to EIA. One such theme is a mathematical model to predict mercury levels in fish tissue in response to contaminant doses.

The following sections present a case for adopting the conceptual model approach in the design and execution of aquatic contaminant impact studies in Canada. The case is presented by:

- Describing the conceptual framework for aquatic contaminant impact studies developed by Hakanson (Section 4).

- Applying the framework approach to a Canadian case study (Lupin Mine aquatic studies) to illustrate study desiyn and data analysis benefits, and carrying out a first order analysis of the Lupin data to generate contamination factors for sediment, clams and fish (Section 5).

- Applying an ecological risk index based on sediment contamination to the Lupin Mine data, other Canadian case studies and Swedish lake contamination studies. This level of analysis has relatively low prognostic capability but it may be valuable to resource managers. The risk index provides a basis for rankiny of contamination problems between water bodies and isolating. specific contaminant parameters of concern (Section 6).

- Demonstrating the application of second order models (i.e. diagnostic/prognostic models) to contaminant impact assessment and aquatic pollution control, describing the status of relevant Canadian data bases and outlining data requirements for further model development (Section 7).

It is hoped that the presentation of this case will contribute to the much needed transfer of scientific knowledge from the academic arena to the applied arena of aquatic pollution control. It is also hoped that this case will demonstrate the benefits of the framework approach for improving the practical and scientific value of aquatic contamination studies in Canada.

4.0 A FRAMEWORK FOR ASSESSMENT OF AQUATIC CONTAMINATION

4.0 A FRAMEWORK FOR ASSESSMENT OF AQUATIC CONTAMINATION

4.1 INTRODUCTION

The conceptual framework (Hakanson 1984A) for assessing the effects of aquatic contamination provides a systematic procedure for tracing contaminants from dose to response, while accounting for the mediating influences of environmental factors. The framework is based on geoecological principles accounting for abiotic factors and processes which govern and interact with biotic components. This basis in geoecology provides a sound rationale for a study design (e.g. sample site selection, sample numbers, standards for collection/analysis) to generate time and space comparable data. In summary, the framework provides a clear, logical system to account for important parameters mediating the behaviour of contaminants in aquatic systems and to describe their relationship to one another.

The components of conceptual framework may be expressed by the following notation:

$$E = f(D, T, W_i) + R$$

where:

E = a parameter expressing an ecological effect (e.g. changes in a valued ecosystem component or an effect parameter directly related to the valued ecosystem component)

D = a contaminant dose parameter (e.g. metal levels in water or sediments)

T = a factor expressing toxicity

W_i = the ith factor or parameter expressing receiving water sensitivity

R = a residual term (the unaccountable remainder; a measure of the model's effectiveness).

The relationship of contaminant dose to environmental response is also influenced by space and time. Thus the effect terms may be transformed to indices by integrating these factors. Using the conceptual framework notation, spatial aspects of the effect may be incorporated as follows:

$$E'' = \int^{A} E' \quad \text{or} \quad E'' = A \cdot E' + R$$

The effect term (E) is transformed into an effect index (E') to allow for quantitative comparison of different effect parameters, e.g. mercury content in fish muscle with benthic indices, and then integrated over the "biological contact area" (A). In this case E" stands for the potential ecological risk accounting for "biological contact area". The greater the area affected by aquatic contamination, the greater the potential ecological risk.

Temporal aspects of the contaminant dose may be accounted for by "biological contact time" (P) which describes the duration of a given contamination event and is based on the assumption that the ecological effect varies directly with the duration of the contamination (under otherwise comparable conditions). Integration over time yields:

$$E''' = \int^{P}\!\!\int^{A} E' \quad \text{or} \quad E''' = P \cdot A \cdot E' + R$$

where E''' stands for the potential ecological risk accounting for time and area.

The "additive effects" of different toxic substances polluting the same receiving water may be incorporated by an overall index accounting for the sum of the effects of the individual pollutants as follows:

$$PER = \sum E''' + R \quad \text{or} \quad PER = \sum P \cdot A \cdot E' + R$$

where PER is the potential ecological risk index for a given set of contaminants in a given receiving water. While recognizing effects may not be strictly additive this provides at least a starting point to direct further assessment.

The conceptual framework is illustrated schematically in Figure 3 with some relevant examples for the various components of the framework. Further discussion on each of the framework components is presented below.

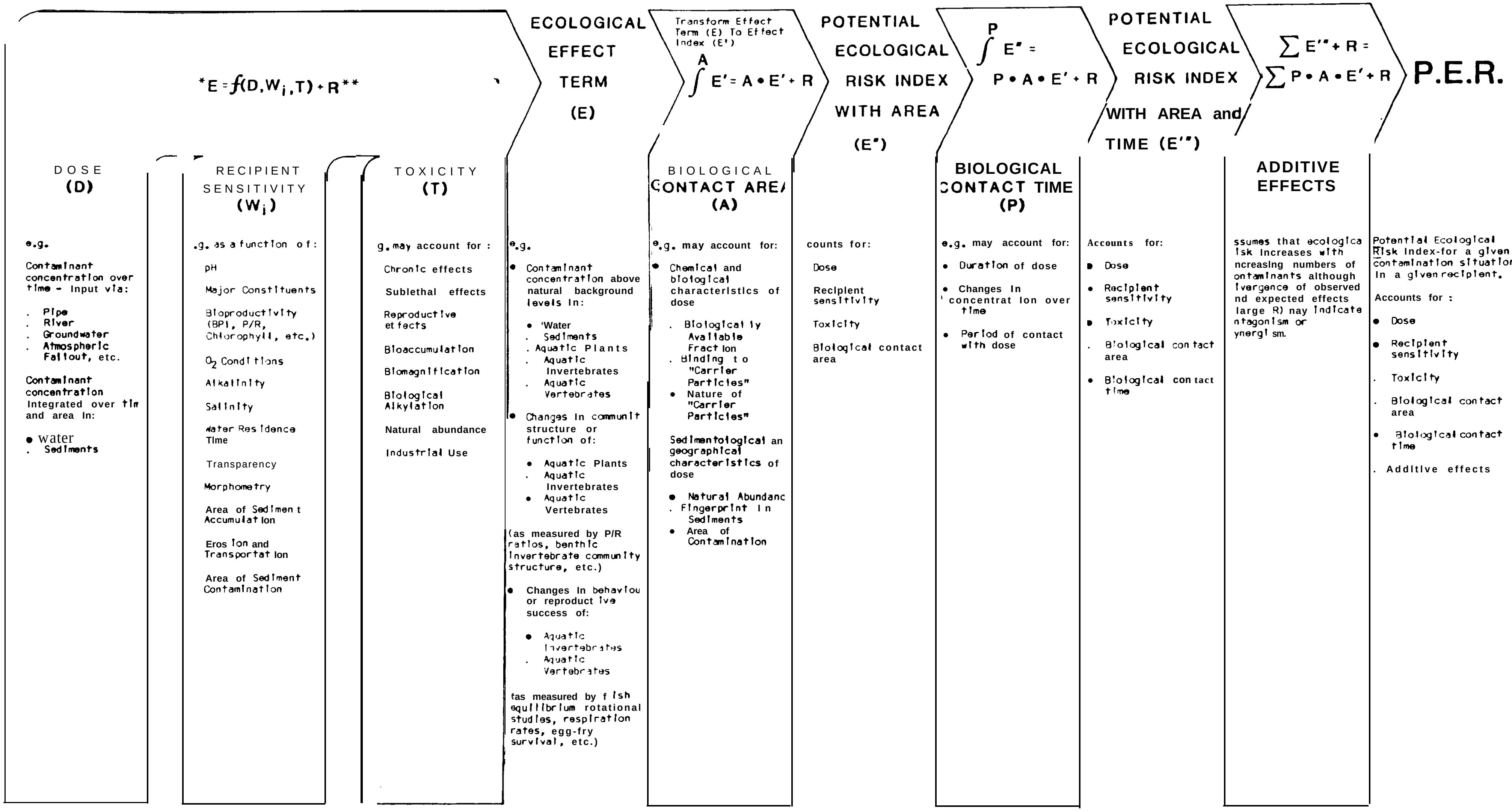
4.2 THE EFFECT PARAMETER

In aquatic contaminant impact studies it is not possible to examine all components of the aquatic ecosystem potentially affected by a contaminant dose. Representative or critical effect parameters (E) must be selected which are applicable to impact assessment and scientifically rationale.

From the standpoint of impact assessment it is important that the effect terms include some measure of valued ecosystem components as described by Beanlands and Uuinker (1983). These are the potential effects which are viewed as most important by the public and resource managers (e.g. the effect on quality of a fisheries resource expressed as metal concentrations in fish tissue).

The selected effect terms also should represent as large a proportion of the potential "effect field" as possible. This may be

Figure 3. Schematic of the conceptual framework with examples of parameters which could be used for each component of the framework depending on the nature of the contaminant and the resource management objectives.



- * This conceptual relationship provides a basis for the development of prognostic models of ecological effect.

- * (R) Residual Ten measures effectiveness of the prognostic model in accounting for principal cause-effect linkages and natural variability in the ecological effect term.

accomplished, for example, by selecting effect terms which integrate a number of sub-effects (e.g. a tertiary consumer). In some instances, the valued ecosystem component may represent only a small proportion of the effect field and so complementary effect terms must be selected. The more fully the effect terms represent the total effect field, the lower the "residual term" will be.

Ideally, the effect terms should also. include the most sensitive or weakest link in the effect field (these may or may not be "valued" ecosystem components) to provide a safety factor or an early warning system to predict responses in the less sensitive effect parameters.

To meet the practical constraints of impact assessment studies, while providing the opportunity to combine and transfer the knowledge gained through individual studies, a limited number of effect parameters should be selected. They should be easy to determine and measurable by standard and easily reproducible methods.

4.3 THE DOSE PARAMETER

The contaminant dose parameter (D) may be measured in various ways. The ideal dose parameter is one which yields integrated information concerning its expression in the receiving environment and which yields the most information with the fewest samples. Hakanson, for example, in his work on contaminated aquatic systems, suggests accumulation-bottom sediments, sediment trap data and/or data from aquatic vegetation. In predictive applications, the dose parameter may be defined in part by a simple mass balance equation expressing input flow volumes and concentration of contaminants. It is also necessary, however, to know somethiny about the relationship of the contaminant to its carrier particles (i.e. its fractionation characteristics) and the physical system dynamics which will dictate its expression in the receiving environment.

4.4 THE TOXICITY FACTOR

The toxicity factor (T) accounts for the fact that various contaminants have different toxic effects on aquatic organisms. Usually in aquatic toxicity studies the toxicity factor is accounted for by the results of laboratory bioassay procedures. Toxicity factor in the context of this framework is directed towards measurement of toxicity in natural waters. The toxicity of contaminants may be ranked, for example, on the basis of their natural abundance in the receiving environment. Use of these relationships provides relevant dimensions to contaminant problems and helps in the selection of appropriate effect terms.

4.5 THE SENSITIVITY OF THE RECIPIENT

Sensitivity parameters (W_i) are those factors in natural receiving waters (e.g. pH, O_2, alkalinity, salinity, bioproductivity, water exchange) which affect the way in which contaminants associate with carrier particles and the chemical form of the contaminants. These factors influence the distribution of contaminants in receiving environments, and the ecological response to contaminants. As a result, responses of organisms in the natural environment may be different to toxic responses produced by laboratory bioassays. Accordingly, sensitivity factors should be accounted for in the assessment framework and they should be measurable by simple, accepted, standardized means.

4.6 THE RESIDUAL TERM

The residual term (R) represents natural variability and environmental factors, unaccounted for in our study design, which give rise to a difference between predicted and observed responses of aquatic systems to a contaminant dose. Theoretically, if we had sufficient knowledge of the basic mechanisms acting on our dose-response pathway and could quantify all causal relationships between dose, sensitivity and response, we could reduce the R term to a minimum representing only sampling variability. On the basis of our present knowledge of ecology, this possibility is remote and the practical constraints of impact studies preclude comprehensive modelling of all possible environmental factors. Furthermore, such fine resolution is not necessarily required for effective environmental management. Thus, the objective of the framework is to express quantitatively (by theoretical, empirical, intuitive or mathematical models) normative E-values from a limited number of readily measurable and representative integrating variables and to minimize the residual term. The R term provides us with a measure of our success in accounting for the major operative mechanisms and the natural variability of the studied ecosystem.

4.7 BIOLOGICAL CONTACT AREA

The biological contact area (A) provides an important dimension to the dose term which permits ranking of impacts. This parameter incorporates two elements: the geographic areas of influence of contaminant dose; and the biologically available fraction of the contaminant dose.

The physical area of influence of a contaminant may be described by standardized methods; for example, the distribution of contaminants in sediments as a function of bottom dynamics. An understanding of the physical dynamic processes (e.g. those giving rise to erosion, transportation and accumulation zones) is a prerequisite to the definition of this parameter.

Biological contact area, as a function of the biologically available dose, provides finer resolution to the ecological risk or impact assessment. Implementation of a simple standardized fractionation scheme, yielding, for example, exchangeable, organic and inert fractions of contaminant dose can provide valuable information to risk assessment.

4.8 BIOLOGICAL CONTACT TIME

The complete framework should also account for the relationship between duration of a contaminant dose and ecological risk. Biological contact time (P) is in turn linked to the dose characteristics (concentration), the contact area and the duration of interface with effect parameters. The framework assumes that the potential ecological effect increases with increased biological contact time (under otherwise comparable conditions) and vice versa.

4.9 ADDITIVE EFFECTS

Since different toxic substances can pollute the same recipient, it is necessary to strategically prepare for additive effects in the framework. The additive effects term accounts for the potential increased ecological risk arising from the impact of numerous contaminants discharged to a single recipient water body. In fact, the impact of more than one contaminant may involve principles of antagonism and synergism, which are very interesting, complex and little studied phenomena in natural aquatic environments. Thus, while effects may not be strictly additive, the framework at least provides a starting point from which these concepts can be further addressed.

4.10 APPLICATION OF THE FRAMEWORK

The framework is not intended to provide a rigid format which must be closely adhered to in every study of aquatic contamination. It is more a systematic notation system derived from our current practical understanding of environmental impact and, specifically, aquatic contaminant impact relationships. It is a skeleton which provides a common denominator for communication amongst impact assessment practitioners and it is meant to be expanded, revised and adapted as our understanding of impact assessment grows.

Subsequent sections of this report will demonstrate with schematics how elements of this framework have been extracted, combined and developed for specific assessment applications. We hope to illustrate how the use of a framework approach can generate practical tools for aquatic pollution control while improving our scientific knowledge of aquatic contaminant systems.

5.0 USING THE FRAMEWORK APPROACH - A CANADIAN CASE STUDY

5.0 USING THE FRAMEWORK APPROACH - A CANADIAN CASE STUDY

5.1 INTRODUCTION

This section of the report demonstrates the application of the conceptual framework approach to a current Canadian case study, the Lupin Mine aquatic studies.

The Lupin studies were not originally designed using the framework approach described in Section 4. The object of this case study analysis was to critically review the Lupin studies using the framework as a template to identify areas where the value of such studies could be increased relative to the effort expended.

This section begins with a description of the context, rationale and components of the original Lupin studies followed by the framework analysis and recommendations for areas of improvement.

5.2 THE LUPIN STUDIES

Detailed documentation of the concurrent Aquatic Studies and Waste Management Compliance Studies for Lupin Mine is contained in separate reports by Reid Crowther & Partners Ltd. (1985A and B, respectively).

As indicated by the two reports, the dose and effect components of the Lupin aquatic contaminant investigations were examined under separate terms of reference with different objectives. The dose, or waste management, studies were aimed at characterizing the effluent and designing a waste treatment process to achieve predetermined effluent standards. The effects, or aquatic, studies were aimed primarily at generating baseline data for future effects monitoring. Although outside of the original terms of reference, some of the information generated by both studies was inevitably integrated and interpreted in an impact assessment context (Reid Crowther & Partners Ltd. 19858).

The effects parameters examined in the Lupin aquatic studies were selected by the consultants through discussion with technical specialists representing environmental management agencies on the Technical Advisory Committee of the Northwest Territories Water Board. As little was known about the recipient water bodies, the studies had two general objectives: to describe the fisheries resource (species, numbers, habitat use) potentially affected by tailings decant effluent; and to gather baseline data on environmental parameters (water, sediments, benthic invertebrates, fish tissue) which would be measured in an effects monitoring program during mine operations.

Without knowledge of what the tailings pond decant discharge rate ultimately would be, it was determined that the area of likely short term effects would be the Seep Creek drainage pathway and Inner Sun Bay on Contwoyto Lake. It was assumed that contaminants would rapidly disperse in Outer Sun Bay with potential gradual long term increases in sediment contaminant levels in the Outer Bay. Thus, monitoring sites were distributed along the effluent pathway from immediately downstream of the tailings impoundment to Outer Sun Bay. The contaminants of concern were assumed to be the metals (e.y. arsenic, copper, lead, nickel, zinc) and possibly cyanide and cyanide complexes. The likely effects of concern were considered to be potential chronic toxic effects and metal accumulation in aquatic organisms.

The parameters selected for study during the pre-discharge stage and the rationale for measuriny them were as follows:

- Water Quality - was selected as a parameter which would permit quantitative measurement of change, either long term (if the effluent was to be discharged continuously throughout the open water season) or short term (if the effluent was discharged during a short period on an intermittent basis). Sites were located at the outlet of each small lake along Seep Creek, at the mouths of Seep Creek and Concession Creek and in Sun Bay to allow mass balance calculations of contaminant. inputs along the length of the effluent pathway. Seasonal measurements were made over three years to characterize natural variability. Samples were analysed for a comprehensive series of chemical and physical parameters, and considerable emphasis was placed on quantifying low metals concentrations.

- Sediment Quality - was chosen as an integrator of contaminant dose over time and as a paramater for which quantitative data could be generated. Cores of varying depths were collected from basins along the effluent pathway from the impoundment area to Outer Sun Bay. Samples were taken from the deepest part of each lake or in deposition areas indicated by the presence of fines in shallow lakes., 'the finesfraction for each sample was analysed for metals concentrations; 'Particle -size and organic carbon analyses were conducted on whole samples. Seasonal measurements were made over two years to describe natural variability.

- Fish Tissues (liver and muscle) - were collected for metals analyses in conjunction with the general fisheries descriptive study. Fish were considered the most valued aquatic resource of the study area. Sun Bay is a popular fishing spot for mine employees and was identified as a traditional area of Inuit camps. Thus, metals levels in fish tissue were considered an important and quantifiable effect term.

- Benthic Macroinvertebrate Communities - were selected as a linkage between sediment accumulations and fisheries and an as integrator of contaminant effects over time. Community indices were examined as a potential measure of chronic effects. Seasonal measurements were made over three years to characterize natural variability.

During the course of the aquatic studies, various circumstances arose which altered the context of the aquatic studies. These are discussed below.

- A large part of the waste management study was directed towards experimentation to determine the effectiveness of various treatment methods. A treatment system was implemented to meet prescribed effluent standards. Aquatic studies data and water intake monitoring data were used by Echo Bay Mines Ltd. to negotiate a higher effluent concentration for zinc. Tailings pond waters were monitored prior to decant according to licence requirements (i.e. total metals levels were measured). When the tailings liquids failed to meet total arsenic standards, decant release was negotiated on the basis of bioassay tests which sugyested that only a minor fraction of the arsenic in the effluent would be biologically available.

- Requirements for internal modifications in the tailings impound=ment dictated a high decant discharge rate in the first year of decant release. This resulted in the designation of Sun Bay, rather than Seep Creek, as the receiving body and negated the rationale for collecting monitoring data on upper Seep Creek.

- Recommendations which emanated ultimately from the waste management compliance program required supportiny analysis addressing potential environmental effects. Data from the aquatic studies were used to make quantitative predictions of water quality in Sun

Bay, and these in turn were used to make qualitative assessments of potential effects on benthic communities and fisheries.

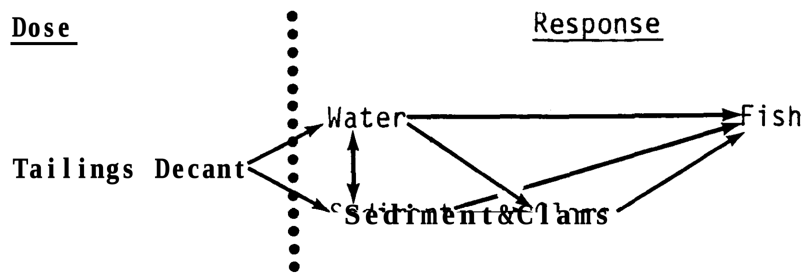
The circumstances surrounding this study, which rendered some data collection extraneous and highlighted the need for other data not collected, arose primarily from the segregation of the dose and response component by separate study objectives. This experience provides a strong justification for using the framework approach at the outset of the study, to ensure that the output from the dose-oriented studies (frequently engineering studies) is compatible with the input requirements of the response-oriented studies (environmental studies).

The following section evaluates the parameters used in the Lupin studies in the context of the conceptual framework and identifies areas needing improvement.

5.3 FRAMEWORK EVALUATION

5.3.1 Study Design

The main premise for the design of the aquatic studies using the conceptual framework is to identify critical linkages between contaminant dose (metals) and environmental response. The idea is not to examine every possible linkage but to define a primary logical pathway which links dose to response and which contains easily measurable effect terms, some of which will have time comparability to the dose. To have time comparability with the dose, the effect term must have a measurable response which is integrated over the duration of the dose and is proportional to that dose. Thus, in order to observe a response in an effect term, the dose term must meet certain criteria with respect to concentration and/or contact time with the effect term. The main pathway for metal contamination examined in the Lupin study is as follows:



Metals in sub-acute toxic concentrations may affect the receiving environment through accumulation, with consequent impacts on the quality of effect terms (e.g. fish tissue) and/or by chronic effects on the physiology of the ecosystem with functional changes (e.g. productivity) occurring in the effect terms. In practically oriented pollution control studies, effect terms must be easily measurable and have some relevance to resource managers. Thus, the pathway examined for the purpose of this case study relates primarily to the accumulation of metals in the receiving environments and particularly in fish tissue.

As noted in the previous section, benthic invertebrate community composition originally was selected as an effect term for the study. Changes measured in this term can provide more insight into the yeneral health of the receiving environment; however, these changes are more difficult to measure and interpret and thus are more difficult to apply in a management context. The Environmental Protection Service of-Environment Canada (EPS) used this effect term in their decant monitoring program; however, their results are not included in this report.

Clams were added as an effect term to improve representation within the potential effect field. The clam data provide a direct link between the dose and the response of concern (metal levels in fish tissue). They are a more sensitive integrator than the sediments of the dose in the transportational environment of Sun Bay. Further, they provide a sensitive link in the contaminant pathway which can show deleterious trends prior to their expression in the effect term of concern (fish tissue) such that corrective steps can be implemented if justified.

The dose (tailings pond decant) was released for approximately one month. In this situation, one would expect a rapid decline of the dose to the water of Sun Bay, via Seep Creek, after the decant siphons were shut down. Water as a response term measures effects over a very short time period, say hours to days, and should respond rapidly to changing concentrations in the dose. The dose to the sediments of Seep Creek, however, would be transported through Seep Creek and released to Sun Bay over a longer time period. Sediments measure effects over long time periods, say years, if they are collected from accumulation areas. Sediments collected from transportational or erosional zones measure effects over unknown time periods. Clams measure effects over a period of months, particularly over the summer growth period, while fish integrate a response over a period of months to years.

Effect terms in Sun Bay were examined during and two months after the decant period. One would expect a response in the water column of Sun Bay during, and for a short period after, decant; a response in the clams within two months of decant; but little or no response in the fish and sediments because of the lack of contact time. Also, the transportational nature of Sun Bay sediments make them a poor choice as an effect term. One way to relate sediment effects in transportational zones to known time periods is with the use of sediment traps which can measure response over a period of months.

It is important that effect terms be expressed in a manner meaningful to resource managers. Using the framework approach, the effects terms in this study may be expressed as metal contamination factors for water, sediment, clams and fish. Contamination factors relate the post-decant concentrations to the natural background concentrations (which contain a statistical measure of variability). Thus, the the contamination factors may be used directly as a measure of the response or effect. They provide a synopsis of the data which, taking into account the time comparability of the response, can provide a basis for decision-making.

Figure 4 takes the schematic outline of the conceptual framework (Figure 3), and illustrates the parameters measured in the Lupin study and opportunities for analysis relevant to impact prediction and resource management. The following sections describe the Lupin study in more detail and identify areas for enhancing information value using the framework approach.

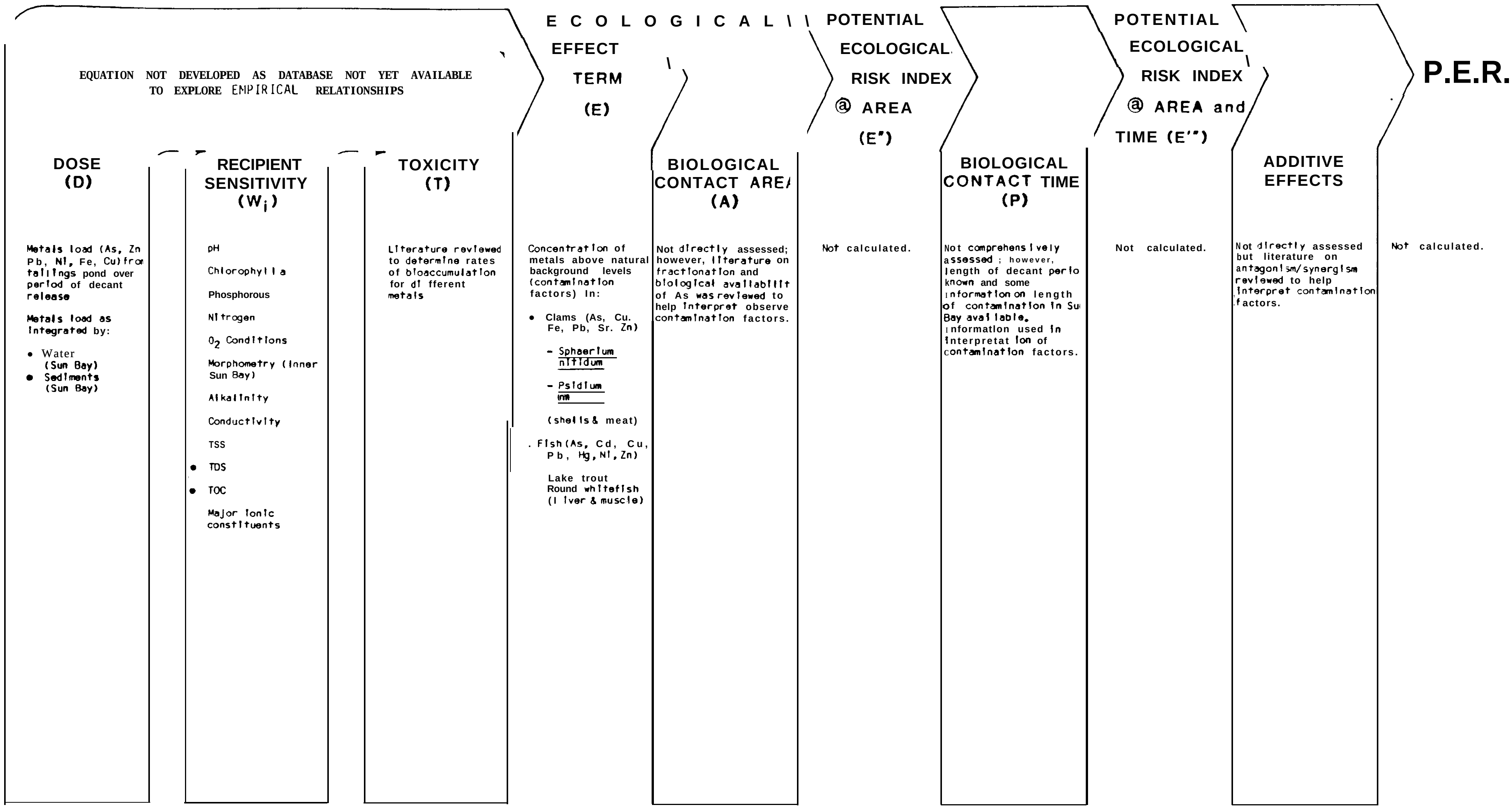
5.3.2 Study Methods

The methods of sample collection and analysis are briefly described to provide some background to subsequent discussion and evaluation of study results. All pre-decant data were collected prior to initiation of this project. The need to use pre-decant data from several sources placed some restrictions on the way post-decant data was collected. The methods employed were as follows:

The Dose Term

The tailings pond decant was monitored by Echo Bay Mines' Environmental Laboratory with periodic crosschecks by a commercial laboratory in Edmonton. Daily flow rates were estimated and reported as m^3/day. Daily grab samples were analyzed for total arsenic (silver diethyldithiocarbamate method), zinc, lead, nickel, iron and copper (atomic absorption spectrophotometry - APHA 1980). The approximate dose (kg) was calculated by multiplying daily flow rates with the metal concentrations and appropriate conversion factors, then summing over the decant period for each metal.

Figure 4. Schematic of the conceptual framework illustrating parameters measured in the Lupin case study and opportunities for impact analysis.



The Response Terms

Water: Water samples collected before and after decant by Reid Crowther personnel were analyzed by Cantest Laboratories, Vancouver. For details on methods see Reid Crowther (1985A) and for sampling locations see Figure 2. Water samples collected during decant period from sample site #925-22 were analyzed by Echo Bay Mines personnel as described previously.

Sediments: Sediment samples prior to decant were collected with a gravity corer, the whole core was sieved and the -53 mm fines were analyzed (Reid Crowther 1985A). For samples collected after decant, only the top 5 cm was sieved and analyzed, so as to be comparable to the EPS study which was taking place concurrently. As the Bay is predominantly transportational and subject to frequent resuspension of sediments, the sediments are very homogeneous in composition thus differences in sediment composition attributable to sampling methods are expected to be minimal. Post-decant sediment sampling was attempted at all water quality sites, but was successful only at EPS Site 3 and EPS Site 6 (Fiyure 2).

Clams: Clams were collected prior to decant by EPS at EPS Site 3 (Figure 2). They were collected using an Eckman dredge and frozen for storage. Clams were collected after the decant period (under ice) by Reid Crowther personnel using a long-handled net at three locations several meters apart at EPS Site 3. Clams and snails were separated from detritus and sediment, then frozen for storage. All clams and snails were shipped to Cleveland State University where samples were thawed, shells and meats were separated and separate pellets of meat and shell suitable for x-ray fluorescence determination were produced. The pellets were then analyzed by x-ray fluorescence techniques developed by Dr. Michael J.S. Tevesz and Dr. Robert L.R. Towns.

Fish: Fish were collected using gillnets at two sites (Figure 2) before and during decant; on the east side of Contwoyto Lake (ESC), and in the narrows of Sun Bay (SB). Fish also were collected from the narrows following decant. The fish were processed and analyzed using methods similar to those outlined in R.L. & L. Environmental Services Ltd. (1985).

Sensitivity Factors

Measurements of pH, alkalinity, dissolved oxygen and bioproductivity were made in conjunction with water and sediment sample analysis (Reid Crowther 1985A). Bioproductivity parameters measured in the water included nutrients and chlorophyll a (APHA 1980); those measured in sediments included organic carbon and nitrogen.

5.3.3 Results and Discussion

The Dose Term

Decant: The tailings pond decant occurred from September 5, 1985 to October 1, 1985 and released a total cumulative volume of approximately 4,414,000 m^3. The daily flow rate with five siphons operating was approximately 200,000 m^3/day or about 2 m^3/s. The actual daily flow rates and metal concentrations in the decant were approximately constant throughout the decant period (Table 1).

The approximate total metals dose released during the decant period ranged from 24 kg for lead to 9,000 kg for iron (Table 2). It should be stressed that the dose concentrations are expressed as total metals and do not necessarily indicate the fraction which is biologically available. The relative order of the total metals dose from lowest to highest was: lead, nickel, copper, zinc, arsenic and iron.

Table 1. Chemical and physical data (from Echo **Bay Mines' Environmental Laboratory) used in calculating metals doses (Site Number** 925-10 **Northwest** Territories Water Board Licence Number N7L3-0925).

Month/Day 1985	Daily Flow Kate (m^3/day)	TOTAL METALS CONCENTRATIONS (mg/L) As	Zn	Pb	Ni	Fe	cu
September							
6	226,000	0.63	0.250	0.013	0.048	2.21	0.113
7	226,000	0.42	0.432	0.009	0.043	2.25	0.104
8	226,000	0.49	0.293	0.007	0.040	2.21	0.098
9	227,000	0.58	0.180	0.007	0.043	2.05	0.104
10	222,000	0.65	0.190	0.005	0.046	2.03	0.103
11	218,000	0.52	0.232	0.005	0.042	1.99	0.102
12	218,000	0.56	0.168	<0.002	0.052	2.01	0.102
13	212,360	0.54	0.175	0.002	0.035	1.92	0.098
14	215,296	0.64	0.165	0.002	0.040	1.89	0.095
15	212,605	0.61	0.188	<0.002	0.038	1.72	0.090
16	210,404	0.63	0.208	0.002	0.038	1.70	0.088
17	208,691	0.68	0.188	<0.002	0.035	2.25	0.088
18	209,425	0.66	0.195	0.008	0.038	2.25	0.088
19	203,064	0.65	0.238	0.002	0.038	2.02	0.085
20	203,064	0.63	0.188	0.002	0.030	2.20	0.082
21	199,883	0.62	0.188	0.002	0.030	2.12	0.082
22	195,969	0.65	0.135	0.002	0.030	2.30	0.080
23	194,256	U.68	0.138	0.002	0.028	2.28	0.080
24	190,586	0.71	0.182	0.022	0.032	1.58	0.078
25	188,140	0.70	0.172	0.002	0.032	1.80	0.072
26	0						
27	0						
28		0.67	0.215	0.013	0.035	1.93	0.078
29	81,470	0.65	0.180	0.008	0.038	1.90	0.073
30	76,822	0.65	0.153	0.018	0.035	1.98	0.073
October							
1	48,472	0.68	0.148	0.013	0.035	2.20	0.073

Table 2. Approximate dose (kg) of total metals released from Echo Bay Mines tailings pond at Site Number 925-10 for the decant period of September 5, 1985 to October 1, 1985.

TOTAL METALS (kg)					
Arsenic (As)	Zinc (Zn)	Lead (Pb)	Nickel (Ni)	Iron (Fe)	Copper (Cu)
2700	905	24	170	9000	400

The Response Terms

Water: During decant, metals concentrations in Inner Sun Bay (Table 3) were approximately one-third of metals concentrations in the tailings decant for the same days (Table 1). This indicates a 3:1 dilution of decant flows in Seep and Concession Creeks. The concentrations of metals in the waters of Inner Sun Bay were subsequently reduced by approximately tenfold to below detection levels of all metals, except zinc, when sampled in November (Table 3). Surface water samples from Outer Sun Bay (EPS 6) and Contwoyto Lake near the mouth of Sun Bay (WQ 11) showed concentrations below detection limits for the metals of concern, while bottom water samples from the same sites showed arsenic concentrations at 0.002 mg/L and < 0.001 mg/L, respectively (Appendix A). Had freeze-up not occurred shortly after the decant periods, it is doubtful that any elevations in metal concentrations would have been observed a month and a half after the decant had stopped. The spring freshet flows, which at their peak could displace the 'volume Inner Sun Bay in l-2 days (Reid Crowther 1985B), would likely return all metals concentrations to background levels.

Sediments: As previously mentioned, sediments from transportational zones (areas of resuspension and movement) are of limited value in aquatic pollution control programs because they integrate effects over an unknown time period. The results are included here primarily to demonstrate how natural background concentrations and contamination factors are calculated and to illustrate the importance of knowing how the chosen parameters integrate the response and the dose. The metal concentrations of Sun Bay sediments over a period of several years (Table 4) varied significantly ($p < 0.001$ to $p < 0.02$) between some dates for all the metals except arsenic. This was without any decant release during these time periods. The natural background concentrations were calculated by the formula listed

Table 3. Total metal concentrations in surface waters of Inner Sun Bay prior to, during and after the decant period, and total metals concentrations at other sites after the decant period. All values reported as mg/L.

Date	As	Cd	Cr	Cu	Fe	Pb	Hg	Ni	Zn
PRIOR TO DECANT									
Inner Sun Bay									
21/9/84 - 12/9/84 (N = 17)	<0.001	<0.0001	<0.001	<0.001	0.14	<0.001	<0.00005	<0.005	<0.010
DURING DECANT[1]									
Inner Sun Bay									
18/9/85	0.18	-		0.022	0.88	<0.002	-	0.011	0.041
21/9/85	0.17	-		0.025	0.75	<0.002	-	0.012	0.061
28/9/85	0.22	-		0.024	0.66	0.001	-	0.018	0.068
AFTER DECANT									
Inner Sun Bay									
22/11/85	0.021	<0.001	<0.001	0.002	0.10	0.001	<0.00005	<0.005	<0.010
Outer Sun Bay									
24/11/85	<0.001	<0.001	<0.001	<0.001	<0.03	<0.001	<0.00005	<0.005	<0.010
Contwoyto Lake									
29/11/85	<0.001	<0.001	<0.001	<0.001	<0.03	<0.001	<0.00005	<0.005	<0.010

1 Data from Echo Bay Mines, Environmental Laboratory Sample Site Number 925-22.

(Table 4) and were rounded off to emphasize that the values are not precise.

Using the natural background metal concentrations (Table 4) and the mean metal concentrations determined after the decant (Table 5), a set of contamination factors was determined for the metals. The contamination factors (Table 5) were generally very low for the sediments, although there is some indication of increased arsenic concentrations. The higher arsenic concentrations occurred in Outer Sun Bay, farthest from the source of pollution. This is understandable if bottom dynamics are considered. The highest sediment metal concentrations would be expected to occur in the closest accumulation area in the path of the decant. Since Outer Sun Bay is deeper, its bottom dynamics would be less transportational than the shallow, narrow Inner Sun Bay. This was borne out by the particle size distribution data which indicated a higher percentage of silt and clay at the Outer Sun Bay site. Inner and possibly Outer Sun Bay are essentially acting as conduits for sediments which are ultimately deposited in the main lake.

Clams and Snails: Metals concentrations in clams and snails were chosen as a biological effect term for several reasons. From previous fish stomach analyses it was determined that plecypods (clams) and gastropods (snails) were used as food by lake trout, lake cisco, and to lesser degree, round whitefish. Secondly, specimens were easy to separate from the detritus and sediment of the sample by sieving and hand-picking. Thirdly, the shells provide an integrated response to contamination over the period of shell growth as metals are incorporated into the shell matrix. Fourthly, the use of the shell precluded the need to purge the animals of contaminating sediments in their digestive tracks. Finally, an appropriate method using x-ray fluorescence was available to analyze the shell material. (Calcium interference inhibits use of atomic absorption spectrophotometry methods on shell materials.)

Table 4. Determination of natural background concentrations of metals in Sun Bay' sediments (ug/g dry substance) before tailings pond decant (from Reid Crowther 1985A).

Date	As	Cd	Cr	cu	Pb	Hg	Ni	Zn
20/09/82	6.38	0.14	35.2	13.8	2.90	0.014	24.5	46.4
	5.94	0.13	33.1	13.4	3.34	0.012	23.6	45.7
	6.29	0.13	30.7	14.0	3.02	0.012	23.8	45.8
10/06/83	6.50	0.20	44.8	28.3	1.75	0.013	28.2	55.7
	7.38	0.15	48.5	52.6	1.50	0.014	26.9	62.9
10/08/83	5.75	0.15	34.8	15.4	3.00	0.015	26.2	42.0
	7.13	0.15	42.1	13.4	3.00	0.012	21.8	40.8
	9.00	0.23	38.8	13.7	2.75	0.014	24.8	53.4
	10.30	0.20	40.9	15.0	3.00	0.011	29.3	51.2
22/09/83	6.00	0.13	38.3	25.2	4.00	0.012	22.9	44.2
	9.38	0.13	37.6	33.3	3.88	0.015	23.0	49.8
	6.50	0.10	38.3	29.3	3.75	0.015	21.8	48.8
	7.00	0.10	38.3	45.3	6.88	0.025	25.2	50.7
12/09/84	8.00	0.05	27.8	21.2	6.00	<0.005	22.2	41.7
	6.38	0.05	30.2	28.3	4.13	0.009	19.1	37.9
	6.75	0.05	32.2	20.1	3.50	<0.005	20.8	38.7
For All Dates								
$\overline{X}$	7.17	0.13	37.0	23.9	3.53	0.012	24.0	47.2
SD	1.34	-0.05	5.5	11.9	1.35	0.005	2.7	6.7
n	16	16	16	26	16	16	16	16
$t_{0.975}$	2.13	2.13	2.13	2.13	2.13	2.13	2.13	2.13
X_n	7.91	0.16	40.0	30.4	4.27	0.015	25.5	50.9
Rounded								
X_n	8	0.2	40	30	4	0.015	25	50

* $X_n = \overline{X} + (t_{0.975} \cdot SD / \sqrt{n-1})$

Table 5. Determination of contamination factors (Cf) from mean ($\overline{X}$) metals concentrations in Sun Bay sediments after tailings pond decant in November 1985 using previously calculated natural background levels (X_n) from Table 4.

Location/Date	As	Cd	Cr	Cu	Pb	Hg	Ni	Zn
Inner Sun Bay, Nov. 22. 1985								
$\overline{X}$ (after decant)	9.25	ND	43.3	17.7	ND	0.012	27.0	45.9
Cf (X/X_n)	1.2		1.1	0.6		0.8	1.1	0.9
Outer Sun Bay, Nov. 24, 1985								
$\overline{X}$ (after decant)	25.0	ND	44.6	17.9	ND	0.013	28.4	48.5
C_f ($\overline{X}/X_n$)	3.1		1.1	0.6		0.9	1.1	1.0
Sun Bay Average, Nov. 22-24, 1985								
$\overline{X}$	17.1	ND	43.9	17.8	ND	0.012	27.7	47.2
C_f ($\overline{X}/X_n$)	2.1		1.1	0.6		0.9	1.1	0.9
X_n (Background from Table 4)	8	0.2	40	30	4	0.015	25	50

ND - Not Detected

The concentration of metals in clam shells indicated that the clams did indeed accumulate metals in their shells (Table 6) and could incorporate some response within a one to three month period of exposure but whether this is a maximum response is not known. Using the limited clam data available from the pre-decant period to determine natural background metal concentrations, and the mean metal concentrations of the clams after decant, contamination factors were determined (Table 7). These contamination factors are highly subjective due to the limited pre-decant data and were calculated primarily for illustrative purposes. The resulting contamination factors indicated an increase of iron, lead and zinc in the clam shells while copper decreased and arsenic remained constant. The absence of a response to the larger arsenic dose could be explained two ways: either the clams were not sensitive to the arsenic or the arsenic was in a totally-bound form unavailable for biological uptake. The decrease in copper concentrations in the shell may have been due to an antagonism with one or more of the other metals, possibly zinc. The relatively high lead contamination factor resulting from the dose sugyested that the clams accumulated lead preferentially and/or that a high proportion of the lead dose was biologically available.

The data on clam meat were very limited. The clam meat was analyzed to test the applicability of the x-ray fluorescence technique to tissue material and to see if clam meats were more responsive on a short-term basis than clam shells. The present data is inconclusive in this regard. The use of freshwater clams, particularly their shells, shows excellent promise as 'a biological monitoring tool for metal contamination. Depending on the species of clam, the time period monitored can range from months to years. On larger, long-lived clam species, shells can be sectioned by rings in appropriate timeframes to provide an historical account of contaminant doses.

Table 6. X-ray florescence determination[1] of metals (mg/g of ash weight) in the meats and shells of composite samples of clams (Sphaerium nitidum and Psidium nitidum) from Inner Sun Bay, Contwoyto Lake, N.W.T. before and after tailings pond decant.

Sample Type	Date	Pellet Weight (mg)	Metal Concentration2 (Detection Limits) - ppm As (5-10)	Cu (10)	Fe (10)	Pb (5-10)	Sr (5)	Zn (5-10)
Clam Meat EPS Yellowknife	Aug/85	38.1	202	2039	955	< 10	< 5	485
Clam Meat	Nov 29/85	37.5	176	1472	4802	144	< 5	538
Dextron Blank (for Binding Meats)	-	35.0	ND	7.2	9.0	ND	ND	ND
Clam Shells EPS Yellowknife	Aug/85	23.4	71	156	775	10	1329	64
		25.4	17	156	782	5	1243	63
Clam Shells Site A	Nov 29/85	27.0	36	35	11700	57	1321	62
		26.6	34	30	13080	64	1447	53
Clam Shells Site B	Nov 29/85	26.1	61	66	12330	120	1433	449
		25.3	75	73	12745	122	1433	484
Clam Shells Site C	Nov 29/85	26.5	33	12	11570	83	1401	87
		24.8	64	12	12020	80	1429	114

1 Determinations done by Dr. Michael J.S. Tevesz (Geology Department) and Dr. Robert L.R. Towns (Chemistry Department) of Cleveland State University.

2 The following metals Mn, Co, Ni, Hg and Cd were below detection limits (10-15 ppm) for all samples.

3 Samples collected by Dave Sutherland and Mark Gordon of EPS Yellowknife.

Table 7. Determination of contamination factors (C_f) for clams from limited data available.

Parameters	Metals (ppm)					
	As	cu	Fe	Pb	Sr	Zn
Xnl, Clam Meat	202	2039	955	< 10	-	485
$\overline{X}$, Clam Meat	176	1471	4802	144	-	538
C_f ($\overline{X}/X_n$) for Meat	0.8	0.7	5	14	-	1.1
Xnl, Clam Shells	50.4	156	778.2	7.6	1286	64
$\overline{X}$, Clam Shells	43.9	37.8	12242	87.5	1411	208
C_f ($\overline{X}/X_n$) for Shells	1	0.2	16	12	1	3

1 X_n was approximated from the mean metal concentrations with no account for variance due to the small sample size ($n < 3$).

Snails also were analyzed after the decant period (Table 8). No pre-decant data were avail able. The data for snail shells showed hiyher levels of arsenic, lead and zinc, and hiyher ratios of these metals to iron, than the clam data. This suggests that snail shells may also be a valuable biomonitoring tool for metal contamination.

Fish: The natural background metals levels for liver and muscle tissue from four fish species from Contwoyto and surrounding lakes and from Inner Sun Bay were determined using several year's data (Tables 9 and 10 and Tables 11 and 12, respectively). For a detailed discussion of the spatial and temporal aspects of fish length on variations in the metal concentrations in the fish samples see R.L. & L. Environmental Services Ltd. (1985). The need to collect fish in similar size and age classes during similar time periods (e.g. late summer/early autumn) is important, although Hakanson (1984B) found the variability of metals to be primarily dependent on the fish contamination factor and not on fish age, weight, species, oryan, metal or lake. He found that fish populations with higher contamination factors require a larger sample number than those with lower contamination factors to obtain a mean metal concentration within a fixed statistical confidence interval.

The background levels for Contwoyto and surrounding lakes were calculated to determine if any major differences would be detected between the whole area and Inner Sun Bay. While minor differences were detected for some metals, the levels were generally very close for both sample sets. This was encouraging as it indicated the possibility of determining regional background levels which could be used to evaluate contamination in other northern lakes.

To determine the fish contamination factors, the mean concentrations of metals in fish from Inner Sun Bay (Table 13; Appendix B) were divided by the natural background levels determined for Inner Sun Bay

Table 8. X-ray florescence determination of metals (mg/g of ash weight) in the meats and shells of the snail (*Valata sincera*) from Inner Sun Bay after tailings pond decant. Samples collected on November 29, 1985.

Sample Type	Pellet Weight (mg)	Metal Concentration[1] (Dection Limits) - ppm As (5-10)	Cu (10)	Fe (10)	Pb (5-10)	Sr (5)	Zn (5-10)
Snail Meats, Sites A,B,C	36.8	239	36	1751	ND	ND	2727
Snail Shells, Site A	25.4	121	21	7175	134	663	1096
Snail Shells, Site B	29.7	262	10	6126	95	592	1140
Snail Shells, Site C	25.4	171	20	8060	274	1078	1474

[1] The following metals Mn, Co, Ni, Hg and Cd were below detection limits (10-15 ppm) for all samples.

Table 9. Determination of background metals levels (ppb wet weight) in muscle and liver of lake trout, round whitefish, lake cisco and Arctic char from Contwoyto Lake and surrounding lakes (data from R.L. & L. Environmental Services Ltd., 1985).

	Muscle					Liver				
Metal	$\overline{X}$ (ppm)	SD	n	$t_{0.975}$	X_n[1] (ppb)	$\overline{X}$ (ppm)	SD	n	$t_{0.975}$	Xn (ppb)
Lake Trout										
As	0.022	0.031	144	1.98	27	0.016	0.015	53	2.01	20
Cd	0.009	0.031	144	1.98	14	0.488	0.150	53	2.01	530
cu	0.591	0.372	144	1.98	652	15.163	8.759	53	2.01	17605
Pb	0.027	0.022	144	1.98	31	0.045	0.031	53	2.01	54
Hg	0.204	0.194	144	1.98	236	0.507	0.505	53	2.01	648
Ni	0.072	0.087	144	1.98	87	0.137	0.087	53	2.01	161
Zn	5.051	2.678	144	1.98	5494	30.152	6.062	53	2.01	31842
Round Whitefish										
As	0.028	0.022	36	2.02	35	0.023	0.022	6	2.57	48
Cd	0.008	0.012	36	2.02	12	0.310	0.166	6	2.57	500
cu	0.575	0.371	36	2.02	701	2.419	1.273	6	2.57	3883
Pb	0.030	0.029	36	2.02	40	0.165	0.276	6	2.57	482
Hg	0.061	0.034	36	2.02	72	0.120	0.067	6	2.57	197
Ni	0.113	0.179	36	2.02	174	0.089	0.091	6	2.57	193
Zn	6.821	4.254	36	2.02	8273	24.894	8.766	6	2.57	34968
Lake Cisco										
As	0.039	0.011	34	2.03	43	0.023	0.008	7	2.45	30
Cd	0.007	0.001	34	2.03	7	0.256	0.092	7	2.45	348
cu	0.689	0.218	34	2.03	766	2.554	0.292	7	2.45	2845
Pb	0.027	0.018	34	2.03	33	0.090	0.094	7	2.45	184
Hg	0.081	0.025	34	2.03	90	0.154	0.031	7	2.45	185
Ni	0.066	0.054	34	2.03	85	0.084	0.032	7	2.45	116
Zn	6.380	1.332	34	2.03	6850	32.012	6.188	7	2.45	38202
Arctic Char										
As	0.021	0.013	29	2.05	26	0.011	0.007	11	2.23	17
Cd	0.006	0.001	29	2.05	6	0.466	0.102	11	2.23	538
cu	0.716	0.612	29	2.05	954	18.635	7.178	11	2.23	23697
Pb	0.033	0.029	29	2.05	44	0.044	0.026	11	2.23	62
Hg	0.056	0.023	29	2.05	65	0.140	0.040	11	2.23	168
Ni	0.261	0.750	29	2.05	552	0.144	0.089	11	2.23	207
Zn	4.613	0.747	29	2.05	4902	31.451	3.174	11	2.23	33689

[1] $X_n = \overline{X} + (t_{0.975} \cdot SD/\sqrt{n-1})$

Table 10. Background metals levels (ppb wet weight) in muscle and liver and the ratios of liver to muscle concentrations for lake trout, round whitefish, lake cisco and Arctic char from Contwoyto Lake and surrounding lakes. (= is used to mean "is approximately" in this table).

Metal	Muscle	Liver	K_{LM}
Lake Trout			
As	27 = 30	20 = 20	0.7
Cd	14 = 15	530 = 530	35
cu	652 = 650	17605 = 18000	28
Pb	31 = 30	54 = 50	2
Hg	236 = 240	648 = 650	3
Ni	87 = 90	161 = 160	2
Zn	5494 = 5500	31842 = 32000	6
Round Whitefish			
As	35 = 35	48 = 50	2
Cd	12 = 10	500 = 500	50
cu	701 = 700	3883 = 3900	6
Pb	40 = 40	482 = 480	12
Hg	72 = 70	197 = 200	3
Ni	174 = 180	193 = 190	1
Zn	8273 = 8300	34068 = 35000	4
Lake Cisco			
As	43 = 40	30 = 30	0.8
Cd	7 = 10	348 = 350	35
cu	766 = 770	2845 = 2800	4
Pb	33 = 30	184 = 180	6
Hg	90 = 90	185 = 190	2
Ni	85 = 85	116 = 120	1
Zn	6850 = 6900	38202 = 38000	6
Artic Char			
As	26 = 30	17 = 20	0.7
Cd	6 = 10	538 = 540	54
cu	954 = 950	23697 = 24000	25
Pb	44 = 45	62 = 60	1
Hg	65 = 65	168 = 170	3
Ni	552 = 550	207 = 210	0.4
Zn	4902 = 4900	33689 = 34000	7

Table 11. **Determination of background metals levels (ppb wet weight) in muscle and liver of lake trout, round whitefish, lake cisco and Arctic** char from Inner Sun **Bay, Contwoyto Lake,** N.W.T. **(data from R.L. & L. Environmental Services Ltd.,** (1985).

	Muscle					**Liver**				
Metal	$\overline{X}$ (ppm)	**SD**	**n**	$t_{0.975}$	X_n[1] **(ppb)**	$\overline{X}$ **(ppm)**	**SD**	**n**	$t_{0.975}$	**Xn** (ppb)
Lake Trout										
As	0.023	0.012	43	2.02	27	0.023	0.018	25	2.06	31
Cd	0.014	0.055	43	2.02	31	0.508	0.127	25	2.06	562
cu	0.507	0.208	43	2.02	571	17.212	9.444	25	2.06	21183
Pb	**0.023**	0.016	43	2.02	28	0.043	0.029	25	2.06	55
Hg	**0.198**	U.136	43	2.02	240	U.527	0.516	25	2.06	74
Ni	**0.061**	0.049	43	2.02	76	0.172	0.089	25	2.06	210
Zn	**4.502**	1.127	43	2.02	4835	31.934	3.915	25	2.06	33580
Round Whitefish										
As	**0.030**	0.022	19	2.09	41	0.028	0.019	3	4.30	80
Cd	**0.006**	0.000	19	2.09	6	0.319	0.131	3	4.30	718
cu	**0.476**	0.104	19	2.09	527	1.805	0.598	3	4.30	3623
Pb	**0.027**	0.024	19	2.09	38	0.284	0.381	3	4.30	1441
Hg	**0.062**	0.036	19	2.09	80	0.085	0.035	3	4.30	191
Ni	**0.071**	0.045	19	2.09	93	0.071	0.037	3	4.30	183
Zn	**5.462**	0.761	14	2.09	5837	21.400	2.545	3	4.30	29138
Lake Cisco										
As	**0.043**	0.009	23	2.07	47	0.024	0.008	6	2.57	33
Cd	**0.007**	0.000	23	2.07	7	0.257	0.101	6	2.57	373
cu	**0.651**	0.199	23	2.07	739	2.491	0.263	6	2.57	2794
Pb	**0.030**	0.020	23	2.07	39	0.103	0.096	6	2.57	213
Hg	**0.077**	0.023	23	2.07	88	0.145	0.024	6	2.57	173
Ni	**0.084**	0.058	23	2.07	109	0.086	0.035	6	2.57	126
Zn	**6.713**	1.157	23	2.07	7223	32.307	6.725	6	2.57	40036
Arctic Char										
As	**0.010**	0.006	8	2.36	15	0.010	0.007	4	3.18	23
Cd	**0.006**	0.000	8	2.36	6	0.560	0.065	4	3.18	680
cu	**1.119**	1.086	8	2.36	2087	18.685	3.570	4	3.18	25239
Pb	**0.019**	0.019	8	2.36	36	0.035	0.013	4	3.18	58
Hg	**0.055**	0.024	8	2.36	77	0.117	0.025	4	3.18	163
Ni	0.732	1.375	8	2.36	1958	0.193	0.107	4	3.18	389
Zn	4.368	0.244	8	2.36	4586	32.432	2.166	4	3.18	36409

[1] $X_n = \overline{X} + (t_{0.975} . SD/\sqrt{n-1})$

Table 12. Background metals levels (ppb wet weight) in muscle and liver and the ratio of liver to muscle concentrations for lake trout, round whitefish, lake cisco and Arctic char from Inner Sun Bay and Contwoyto Lake, N.W.T. (= is used to mean "is approximately" in this table).

Metal	Muscle	Liver	$=K_{LM}$
Lake Trout			
As	27 = 30	31 = 30	1
Cd	31 = 30	562 = 560	19
cu	571 = 570	21183 = 22000	4
Pb	28 = 30	55 = 55	2
Hg	240 = 240	744 = 750	3
Ni	76 = 75	210 = 210	3
Zn	4835 = 4900	33580 = 34000	7
Round Whitefish			
As	41 = 40	80 = 80	2
Cd	6 = 10	718 = 720	72
cu	527 = 530	3623 = 3600	7
Pb	38 = 40	1441 = 1400	35
Hg	80 = 80	191 = 190	2
Ni	93 = 90	183 = 180	2
Zn	5837 = 5900	29138 = 29000	5
Lake Cisco			
AS	47 = 50	33 = 35	0.7
Cd	7 = 10	373 = 370	37
cu	739 = 740	2794 = 2800	4
Pb	39 = 40	213 = 210	5
Hg	88 = 90	173 = 170	2
Ni	109 = 110	126 = 130	1
Zn	7223 = 7200	40036 = 40000	6
Artic Char			
As	15 = 15	23 = 25	2
Cd	6 = 10	680 = 680	68
cu	2087 = 2100	25239 = 25000	12
Pb	36 = 35	58 = 60	2
Hg	77 = 80	163 = 160	2
Ni	1958 = 2000	389 = 400	0.2
Zn	4586 = 4600	36409 = 36000	8

(Table 12). The contamination factors for liver and muscle tissue for lake trout and round whitefish (Table 14) generally indicated little or no contamination had occurred. The exceptions were the livers of round whitefish for the metals arsenic, nickel, zinc and copper.

The contamination noted in the round whitefish livers was likely an artifact due to the small sample size for round whitefish pre-decant data and the analysis of the very small liver of each fish on a "wet weight as received" basis for the post-decant samples. The possibility that round whitefish livers are good accumulators of these metals seems unlikely as there were no corresponding increases in metals in the muscle tissues after decant and the natural background concentration in round whitefish livers did not differ greatly from the livers of other fish species.

The general lack of response exhibited by the fish is what would be expected given the short contact time with the dose, the rapid decline of the dose to near background levels and the mobility and metal avoidance behaviour of fish. The mobility of the fish precludes knowing the length of time any given fish has been exposed to the dose.

<u>Summary:</u> The summarized results of the contamination factor analysis (Table 15) indicate the necessity for time-comparability between the effect and the dose terms and the need to be concerned with the form of the dose (i.e. the biological availability). The lack of effect on the sediments can be explained by the transportational nature of the site. The clams showed a response to iron, lead and zinc but no response to arsenic. The proportionately greater response of clams to a low lead dose and the lack of response to a hundred-fold larger arsenic dose would suggest a substantial difference in biological availability between lead and arsenic in this case. The contamination factors for the fish showed a low or mixed response to the dose

Table 13. Mean concentrations and standard deviations of metal concentrations (wet weight ppb) in lake trout and round whitefish livers and muscle tissues from Contwoyto Lake, N.W.T. after the decant period in the fall of 1985.

Location	Species	Tissue Type	n	As		Cd		Cu		Pb		Hg		Ni		Zn	
				$\overline{X}$	SD	$\overline{X}$	SD	$\overline{X}$	SD	$\overline{X}$	SD	$\overline{X}$	D	$\overline{X}$	SD	$\overline{X}$	SD
East Side of Contwoyto Lake	Lake Trout	Liver	5	5.4	0.6	490	97	11,770	5,930	5.4	0.4	200	161	148	49	31,505	6,110
		Muscle	5	5.4	0.3	4.6	2.0	445	141	5.4	0.4	158	65	71	70	4,890	543
Inner Sun Bay	Lake Trout	Liver	10	26.3	44.4	427	124	21,590	13,820	5.4	0.7	223	44	189	96	33,480	6,870
		Muscle	10	22.0	27.2	5.7	0.5	512	148	16.0	32.7	199	35	32.4	13.7	4,995	820
	Round Whitefish	Liver	5	692	820	257	166	7,664	8,298	5.0	0	172	33	597	375	56,500	34,800
		Muscle	5	23.6	17.8	5.8	0.3	390	48	5.8	0.3	104	30	41.3	26.9	5,900	548

Table 14. Determination of contamination factors (Cf) from the mean metal concentrations of liver and muscle tissues in lake trout and round whitefish for Inner Sun Bay, Contwoyto Lake, N.W.T. during fall 1985.

Species	Tissue Type	Parameter	As	Cd	Cu	Pb	Hg	Ni	Zn
Lake Trout	Liver	$\overline{X}$[1]	23.6	427	21,590	5.4	223	189	33,480
		C_n[2]	30	560	22,000	55	750	210	34,000
		C_f[3]	0.9	0.8	1.0	0.1	0.3	0.9	1.0
	Muscle	$\overline{X}$	22.0	5.7	512	16	199	32.4	4,995
		C_n	30	30	570	30	240	75	1.0
		C_f	0.7	0.2	0.9	0.5	0.8	0.4	1.0
Round Whitefish	Liver4	$\overline{X}$	692	257	7,664	5.0	172	597	56,580
		C_n	80	720	3,600	1,400	190	180	2,900
		C_f	8.7	0.4	2.1	0.0	0.9	3.3	2.0
	Muscle	$\overline{X}$	23.6	5.8	390	5.8	104	41.3	5,900
		C_n	40	10	530	40	90	90	5,900
		C_f	0.6	0.6	0.7	0.1	1.3	0.5	1.0

1 After decant from Table 13.
2 Natural background levels for Inner Sun Bay from Table 12.
3 $Cf = \overline{X}/C_n$.
4 Based on a very limited number of background samples (N=3) and $\overline{X}$ determined on "wet weight as received basis" due to insufficient sample size for moisture determination.

which is understandable due to the short contact time and the mobility of the fish.

5.4 IMPROVEMENTS OFFERED BY THE FRAMEWORK APPROACH

Using the template of the conceptual framework (Figure 3) to examine a typical water licence case study (Figure 4), a number of useful observations can be made.

1. First, environmental regulatory studies represent a valuable source of information upon which to build our understanding of aquatic contaminant effects; however, they are valuable only if the objectives of the study are clearly defined and the study approach (parameters, methods) is fully and clearly rationalized. The benefit of the framework is that it orients the study design towards the practical objectives of the study at the outset. It presents a series of assessment milestones (Figure 3) with examples of corresponding study products (e.g. impact prediction models, effects monitoring, contamination factors and ecological risk indices).

   Initially the Lupin study objectives avoided the issue of impact prediction and were oriented towards regulatory monitoring rather than gathering information which would facilitate active environmental management. When the need to address practical management problems arose, impact prediction was inevitable and new information requirements became evident. Impact prediction and the corresponding monitoring of contaminant pathways can provide a valuable source of information pertinent to environmental management. Use of the framework in project design helps to keep these long term goals and opportunities in mind.

2. The framework stipulates the integration of dose (project design) and response (receiving environment) studies, which are frequent-

Table 15. **Estimated dose of total** metals to Inner **Sun Bay and the corresponding contamination factors for sediments, clams and fish.**

	Approx. Dose	Q-Sediment			Cf-Clams		C_f-Fish		
		Inner	Outer	Whole			Lake Trout		Round Whitefish
	(kg)	Bay	Bay	Bay	Meat	Shell	Muscle	Liver	Muscle
Arsenic	**2700**	**1.2**	**3.1**	**2.1**	**0.8**	**1**	**0.7**	**0.9**	**0.6**
Cadmium		ND	ND	ND	ND	ND	**0.2**	**0.8**	**0.6**
Chromium		1.1	1.1	**1.1**					
Copper	400	0.6	**0.6**	**0.6**	**0.7**	**0.2**	**0.9**	**1.0**	**0.7**
Iron	9000				5	16	-		
Lead	24	ND	ND	ND	14	12	**0.5**	**0.1**	**0.1**
Mercury		**0.8**	**0.9**	**0.9**	ND	ND	**0.8**	**0.3**	**1.3**
Nickel	**170**	**1.1**	**1.1**	**1.1**	ND	ND	**0.4**	**0.9**	**0.5**
Zinc	**905**	**0.9**	**1.0**	**0.9**	1.1	3	**1.0**	**1.0**	**1.0**

ly separated and allocated to engineering and environmental specialists respectively. The framework emphasizes the interdependence of these components in the process of environmental management. Integration of dose and response studies ensures an effective focus on critical study parameters and minimizes wasted effort on extraneous or redundant data collection.

In the Lupin example, belated information on the nature of the dose shifted the aquatic study objectives from an effects monitoring focus to an impact prediction focus. As a result, some of the upstream sample sites were rendered extraneous. As well some important information had not been obtained (e.g. fractionation of the arsenic dose).

3. The recipient sensitivity term of the framework forces rationalization of the sampling sites and parameters selected for study. The sensitivity of the receiving water body depends on many factors starting with the basic physical dynamics of the system. The physical basis for sampling programs frequently is weak in programs designed by biologists. In the Lupin study, this lead to an overemphasis on the sediment linkage in contaminant pathways inInner Sun Bay.

   Frequently, far more parameters for recipient sensitivity are measured than will ever be usefully applied to impact prediction or interpretation. Selection of these parameters and frequency of their measurement should be carefuly rationalized in terms of their relationship to the dose parameters and the study objectives. By keeping the intended application of these data in clear view, the investigator is encouraged to work with his data and refine his sampling program or drop parameters which fail to yield instructive relationships.

The framework introduces the potential for developing useful prognostic tools by exploring empirical relationships between dose, recipient sensitivity and effects (Section 7). Developing hypotheses for these relationships provides a rational basis for selecting sensitivity parameters and developing and exploring data bases for these parameters. This ongoing evaluation of the data base provides a check on extraneous data collection. Again, it must be stressed that this approach emphasizes maximum information return for minimum sampling effort and the development of practical (working) tools for contaminant management.

When contaminant studies are oriented towards baseline data collection for regulatory purposes, as was the case for Lupin, there is little impetus to explore relationships between effect and sensitivity parameters or to compare these relationships with other available data bases. Too frequently, the product of baseline studies is volumes of data, very little of which are ever applied to enlarging our understanding of contaminant impacts.

4. Selection of the ecological effect terms in the context of the framework emphasizes how well the chosen parameters represent the impact of the dose in the environment and whether the effects are expressed in terms which are meaningful to resource managers.

   The effect terms initially chosen for the Lupin studies were water, sediments, invertebrate community structure and fish tissue metal concentrations. When it became apparent that the dose would be intermittent and of relatively short duration, the framework was used to evaluate the chosen effect terms. Because of the transportational character of the sediments, the mobility of the fish and the natural temporal variability of invertebrate communities, it became clear that the only term likely to demonstrate an effect in this case was the water. Knowledge of short term change in water quality at sub-acute toxic contaminant concentrations is not particularly helpful to the resource manager whose major concern may be fisheries. Therefore, metal accumulation in molluscs (clams and snails) was added as an effect term which was more apt to integrate an effect over a short time period and represent a link in the contaminant pathway (as food) to the fish. There is equal justification for dropping the sediment analysis as an effect term except in accumulation areas, possibly the accumulation basin in Contwoyto Lake nearest the outlet of Sun Bay. This refinement would result in more information gained for effort expended and would provide a measure of effect at a more sensitive link in the path. If this effect persisted it might ultimately be related to metal contamination in fish.

Hakanson has developed some tools for expressing effects (e.g. contamination factors) which, if widely adopted,could be used to compare the behaviour of contaminants in different systems. Accumulation of such comparative information would ultimately be very valuable to resource managers. This is only one example of many such analytical tools which could be developed for wide use by resource managers.

5. As shown in Figure 4, the Lupin study program did not explicitly address the toxicity, contact area, contact time and additive effects components of the framework, although systematic consideration of these factors helped in data interpretation. By using a framework which incorporates these components, the investigator is encouraged not only to address them in his study design but to seek relevant information from the dose component of the study. While it is generally acknowledged that temporal, spatial and cumulative effects components are critical elements

of any ecological impact study, using the framework in study design will improve the chances that they are systematically considered.

In summary, the framework incorporates fundamental and widely recognized principals of ecological and impact assessment study design. Although these principals have been discussed at length in the impact assessment literature, they have yet to become routinely incorporated in applied aquatic contaminant studies. The framework is a valuable practical aid which can be readily implemented to improve study design and its format encourages further development and refinement. For practitioners of aquatic contaminant impact assessment, the framework provides a valid starting point for enhancing the quality and end product of EIA studies.

6.0 DEVELOPMENT OF AN ECOLOGICAL RISK INDEX FOR AQUATIC POLLUTION CONTROL AND ITS POTENTIAL APPLICATION IN CANADA

6.0 DEVELOPMENT OF AN ECOLOGICAL RISK INDEX FOR AQUATIC POLLUTION CONTROL AND ITS POTENTIAL APPLICATION IN CANADA

6.1 INTRODUCTION

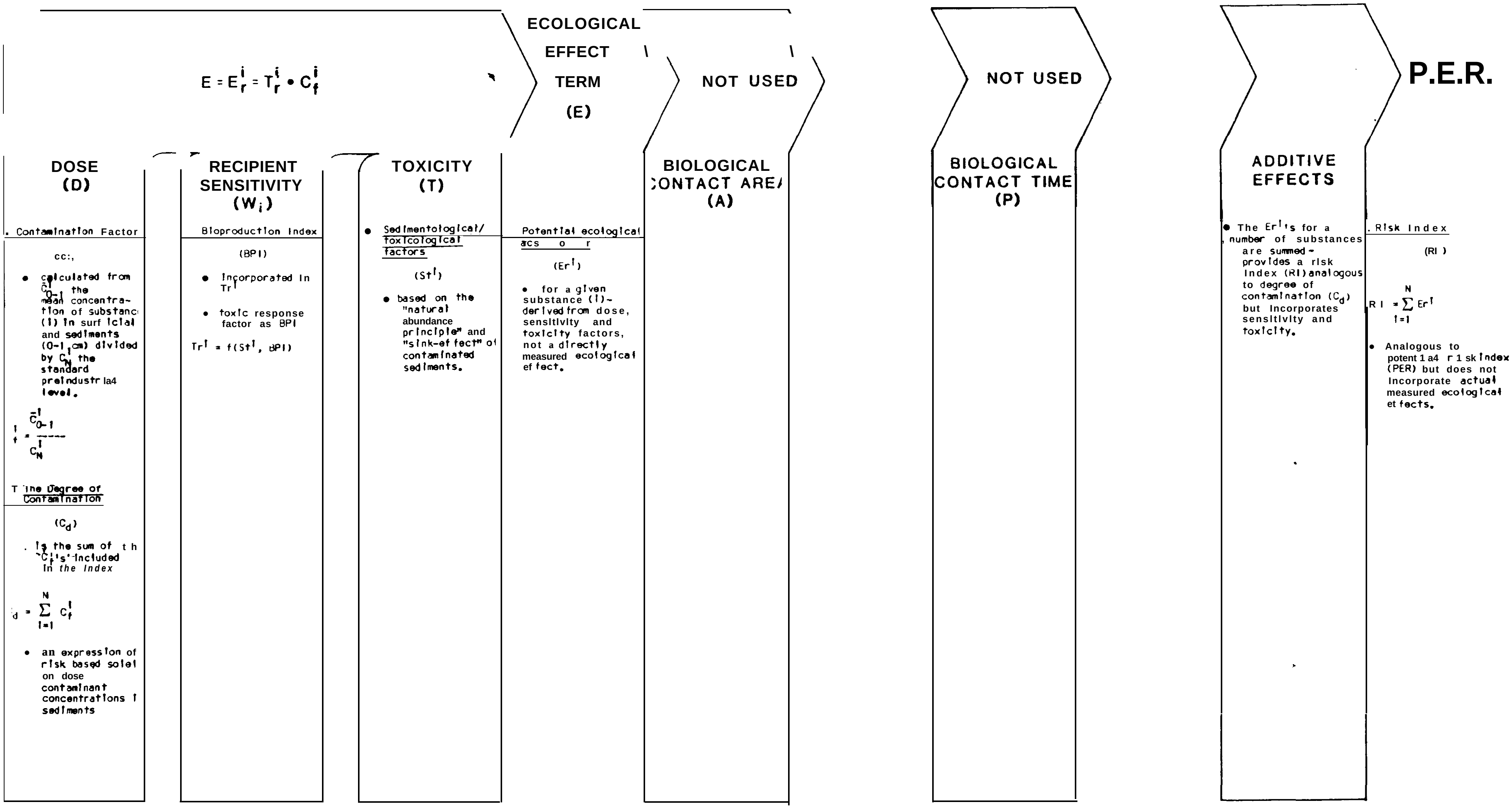
The framework to date has generated a number of analytical tools designed to aid resource managers. One such tool is the ecological risk index, a diagnostic tool for giving relative dimensions to aquatic contaminant problems in a number of lakes. Specifically, the risk index ranks lakes/basins and contaminant substances by degree of ecological risk as a basis for focusing manayement action.

The objective of this section is to demonstrate a way in which data from aquatic contaminant studies may be assembled, analyzed and expressed in terms which are relevant to resource managers. Risk indices are developed using data from a series of Canadian lake contaminant studies and the lakes are ranked according to the degree of ecological risk presented by their respective contamination problems. This analytical approach is fully discussed in Hakanson (1980A) and is presented here in synoptic form along with Swedish and Canadian examples. The use of existing data from Canadian studies to develop risk indices reveals opportunities to improve and standardize our data gathering techniques such that the data generated may be more readily adaptable to resource management applications.

6.2 DERIVATION OF THE RISK INDEX

The risk index (RI) is based on components of the framework (Fiyure 5). Relative risk in a receiving waterbody can be expressed in various ways depending on the information incorporated in the expression (Figure 5). For example, relative risk may be expressed in terms of the dose only, as degree of contamination; as a potential ecological risk factor, incorporating elements of dose, recipient

Figure 5. Schematic of the conceptual framework illustrating the components and terms used in developing a potential ecological risk index for aquatic contamination in lakes.



sensitivity and toxicity; and as a risk index, incorporating the additive effects of several contaminants enteriny the same waterbody. Clearly there are potentially many more ways of expressing relative risk by incorporating more or different components of the framework or by altering the expression of the terms within each component. The ways need only be limited by the objectives of the investigator and/or the availability of relevant data. The following discussion describes one way in which risk indices may be developed using the types of data which are frequently collected in contaminant studies.

This risk index is based on information derived primarily from the sediments and provides an assessment of "potential" ecological risk rather than a direct description of ecological effects as no biological parameters are included in the index. The work is based on the premise that a sedimentoloyical risk index for toxic substances in limnic ecosystems should account, at least, for the following four components of the framework:

- the dose (as integrated by concentration in sediment);
- the sensitivity of the receiving water body;
- the toxicity of the pollutants; and,
- the additive effects (number of pollutants).

This risk index is meant to provide a fast and simple quantitative value expressing the potential ecological risk of a given contamination situation in a given freshwater lake. For the purpose of accuracy, simplicity and rapidity, the risk index uses sediment data. Sediment data, when collected properly, are time-integrated and time-stable when compared to water chemistry data; sediment samples are relatively easy to collect; sample representativeness in time and space can be evaluated; and the generally higher contaminant concentrations in sediments, relative to water, permit yreater analytical accuracy at a lower cost.

The risk index can reflect the threat towards resources valued by man due to increased exposure of these aquatic resources, such as fish, to toxic substances in sediments. Secondarily, the risk index can reflect the potential ecological hazard of a contaminant in a broader biological context (i.e. the risk of destroying the "weakest links in an ecological chain" in a given lake).

The following sections describe the rationale for derivation of risk indices as illustrated schematically in Figure 5.

6.2.1 Concentration Requirement

The concentration requirement addresses the direct relationship between the risk index-value and sediment contamination. In order to obtain a representative measurement of contaminant concentrations in whole lakes or lake basins, the following problems must be overcome:

(a) The problem of lake bottom dynamics - To identify representative sample sites, the physical dynamics of the lake bottom should be characterized (i.e. erosion - transportation-accumulation bottoms), (see Hakanson 1977A,B). Accumulation areas are areas where fine material (medium silt and finer) is being deposited continuously; in transportation areas fine material is deposited discontinuously (i.e. periods of accumulation and transportation are alternating); and, erosion areas are areas where no fine material is deposited. Data from transportation and erosion zones are not suited to development of a risk index because the sediments may be old, complex, highly variable and, thus, impossible to interpret. Consequently, sediment contaminant samples must be taken from accumulation areas. Conversely, one must avoid sampling sediments for concentration measurements in the following environments: rivers and other high energy environments; between islands where a bottleneck effect may be apparent; areas close to river mouths; and areas on sub-aquatic slopes with inclinations greater than 4-5%, where fine material is easily resuspended.

(b) The problem of defining natural background concentrations - This is a central question in all projects dealing with lake sediments as indicators of pollution. The problem can be approached in two quite different ways. One way is to develop a general geological reference level as a standard value for all comparisons. Another way is to determine a preindustrial level for every sediment core using certain pollen horizons. In the first case, all local variations are ignored; in the second case all local differences are emphasized.

Using defined, standardized, preindustrial reference values saves time and money and provides practical and administrative advantages connected with the use of general reference values; once these have been determined, no sampling or analysis of sediments other than surficial deposits from accumulation areas is required. The main disadvantage is lower resolution. For the development and application of a risk index, the first approach is considered most useful and relevant.

What standard preindustrial-reference values should be used and what substances should be included in the risk index? The second question will be addressed in the "number requirements" Section (6.2.2). With respect to the first question, Hakanson determined standard preindustrial reference values from uncontaminated sediments of 50 European and American lakes of varying size, geographical position, trophic level and other limnological characteristics. He defined the standard preindustrial reference level (C_n^i) for each substance as the mean plus one standard deviation, rounded to emphasize that the values are not precise. This definition takes into account the variability of sediment data between lakes; if the data display a low degree of scatter then (X + SD) will be close to X; if a great spread exists this is accounted for in a statistically definable and relevant way. The values in ppm are:

- PCB = 0.01
- Hg = 0.25
- Cd = 1.0
- As = 15
- cu = 50
- Pb = 70
- Cr = 90
- Zn = 175

These values represent the "upper limit" for natural background or preindustrial sediment concentrations.

(c) The thickness of the sediment layer to be sampled and analyzed - This is important because:

- it is difficult to collect good samples if thin layers are utilized (< 1 cm); and

- thick layers provide less accurate resolution in time.

While from a theoretical viewpoint it would be preferable to collect, analyze and compare sediment data from a known and comparable time span, this would require considerable monitoriny and detailed work. As a general monitoring tool, the 0-1 cm layer should be used as a standard.

(d) The number of samples needed to obtain valid mean values - Hakanson (1980A) recommends at least five samples, taken from accumulation bottom deposits and providing even area coverage of a lake .or sub-basin, to make a proper estimate of the mean value.

Bearing these points in mind, the concentration of contaminants can be expressed as a contamination factor (C_f^i) for each element of concern, i.e.:

$$C_f^i = \frac{C^i_{0-1}}{C_n^i}$$

where C^i_{0-1} is the mean concentration in surficial sediments from accumulation areas and Cni is the preindustrial reference level for a given substance, i.

The following classification is used in this risk index approach:

$C_f^i < 1$ => low contamination factor;
$1 \leq C_f^i < 3$ => moderate contamination factor;
$3 \leq C_f^i < 6$ => considerable contamination factor;
$C_f^i \geq 6$ => very high contamination factor.

6.2.2 Number Requirement

The number requirement addresses the assumption that, under similar situations, a lake polluted by many substances should be attributed a higher risk index than a lake contaminated by fewer substances. It is neither practical nor desirable that the risk index incorporate all possible toxic substances. It should always be based on the same parameters for comparative purposes. The question is what substances?

The substances chosen should be representative of groups of substances which occur in the sedirnents in a similar way. Elements such as Fe, Mn and P are unsuitable because their appearance in sediments is often governed by physical/chemical processes in the sediments which cannot be linked unambiguously to contamination. These elements often show very complex sedimentological distribution patterns. Extremely rare elements (e.g. Atomic Number > 50) and substances which cannot be analyzed in a standard manner should also be avoided. Major elements (Si, Al, K, Na and Mg which make up the largest group of the sediment matrix), carbonate elements (Ca, Mg which constitute the second largest group - 15%) and nutrient elements (org. C, N and P - 10%) should be disregarded in a risk index which focuses on toxic substances.

There are, of course, many criteria one can use in the choice of parameters, and the subsequent list should be considered simply as an exarnple. The eight parameters used in the present study were Hg, Cd, Pb, Cu, Zn, As, Cr and PCB. Three co-parameters should also be measured. To determine the physical status of the surficial sediments, the water content (W_{0-1}) must be determined, and to account for sensitivity factors related to bioproduction, nitrogen content (N) and the ignition loss or organic content (IG) can easily be determined.

The degree of contamination (cd) which accounts for the number of contaminants affecting a given waterbody in a quantifiable way, is defined as the sum of the contamination factors (C^i_f). For the eight representative parameters identified above, the degree of contamination would be expressed as follows:

$$Cd = \sum_{i=1}^{8} C_f^i$$

The values C_f^i and Cd may be used to give a standardized description of sediment contamination and are a first step towards a risk index. Note that the contamination factor in itself can provide a "first cut" assessment of relative risk in waterbodies.

The following classification may be used to describe the degree of contamination (C_d value) based on eight parameters:

$C_d < 8$ => low degree of contamination;

$8 \leq C_d < 16$ => moderate degree of contamination;

$16 \leq C_d < 32$ => considerable degree of contamination;

$C_d \geq 32$ => very high degree of contamination; indicating serious anthropogenic pollution.

6.2.3 Toxic Factor Requirement

The toxic factor requirement accounts for the fact that different substances have different toxic effects in aquatic systems. The principles which were used to incorporate this factor are as follows:

(a) The "abundance principle" states that a proportionality exists between toxicity and rarity. The "abundance number" has been determined from concentrations in igneous rock, soils, freshwater, land plants and land animals. The following order of natural abundance has been established between the metals:

Zn	<	Cu	<	Pb	<	Cr	<	As	<	Cd	<	Hg
(1.0)		(3.4)		(13)		(140)		(140)		(230)		(1160)

(b) The principle of "sink-effect" states that different elements will make different "fingerprints" in lake sediments. A "sink-factor" has been determined from the quotient between the natural background concentrations in freshwater for the metals and the preindustrial reference values for lake sediments. The following order has been established:

Cr	<	As	<	Zn	<	Pb	<	Cd	=	Cu	<	Hg
(2)		(27)		(57)		(71)		(200)		(200)		(320)

(c) A "sediment toxic factor" (St^i) expressing the toxicity of an element and its expression in the sediments, can be derived by multiplying the "abundance" by the "sink-factor" and subsequently normalizing to fit the resultant values within a workable range while maintaining their appropriate distances. The Sti value is a constant for each metal and is analogous to the preindustrial reference level (Cni). The values obtained by Hakanson (1980A) are as follows:

Zn=1 < Cr=2 < Cu=Pb=5 < As=10 < Cd=30 < Hg=PCB=40.

6.2.4 Sensitivity Requirement

Different lakes/basins have different sensitivities to different toxic substances. Values for "sensitivity factors" are derived from the same samples used to determine the concentrations of the toxic substances (sediments in the present context). One such "sensitivity factor" is the bioproduction index (BPI-value). The BPI-value is determined from data on nitrogen content and organic content (ignition loss = IG) in the sediments and is defined as the N-content on the regression line that corresponds to IG = 10% (Hakanson 1984C). Using a BPI-value of 5.0, which is characteristic for moderately eutrophic and bioproductive waters, a "toxic-response factor" (Tri) was developed (Figure 5). The "toxic-response factor", which accounts for the "sedimentological toxic factor" (St^i) and the sensitivity requirement (as measured by BPI), has been defined in the following way for the actual substances (Hakanson 1980A):

Substance	St^i-value	Tr^i-value
PCB	40	$40 \cdot BPI/5$
Hg	40	$40 \cdot 5/BPI$
Cd	30	$30 \cdot \sqrt{5} / \sqrt{BPI}$
As	10	10
Pb	5	$5 \cdot \sqrt{5} / \sqrt{BPI}$
Cu	5	$5 \cdot \sqrt{5} / \sqrt{BPI}$
Cr	2	$2 \cdot \sqrt{5} / \sqrt{BPI}$
Zn	1	$1 \cdot \sqrt{5} / \sqrt{BPI}$

6.2.5 The Risk Factor

To quanitatively express the potential risk of a given contaminant in a given lake, we may define the risk factor (Ed) accordingly:

$$Er^i = Tr^i \cdot C_f^i$$

where

Tr^i = the toxic-response factor for a given substance and given lake;

C_f^i = the contamination factor for the given substance.

The following terminology may be used to describe the risk factor:

Table 16. Lake area, maximum depths and source of data for the Canadian lakes examined.

Lake	Province/ Territory	Lake Area (km^2)	Maximum Depth (m)	Source
Rice	Manitoba	3.82	5.1	Beck (1984A)
Snow	Manitoba	5.26	17.4	Beck (1984B)
Herblet	Manitoba	30.1		Beck (1984B)
Trout (Embury)	Manitoba	9*	35"	Wilson (1984)
Cliff	Manitoba	2*	18"	Wilson (1984)
Gods	Manitoba	1050	74.3	Beck (1982)
Francois	B.C.	420	245	Kelso & Jones (1983)
Meg	N.W.T.	0.15	1.5	Moore et al. (1979)
Keg	N.W.T.	0.35	2.5	Moore et al. (197Y)
Peg	N.W.T	0.1	2.0	Moore et al. (1979)
Narrow	N.W.T	0.07	3.0	Moore et al. (1979)
Great Slave	N.W.T.	28,570	157	Moore et al. (1979)
Kootenay	B.C.	389	154	Daley, et al. (1981) Crozier & Duncan (1985)
Sun Bay (Contwoyoto)	N.W.T	8.9	>10	Reid Crowther (1985)

* Estimated value.

$E_r^i < 40$ => low potential ecological risk;

$40 \le E_r^i < 80$ => moderate potential ecological risk;

$80 \le E_r^i < 160$ => considerable potential ecological risk;

$160 \le E_r^i < 320$ => hiyh potential ecological risk;

$E_r^i \ge 320$ => very high potential ecological risk for the substance in question.

6.2.6 The Risk Index

Analogous to contamination factor (C_f^i) and the degree of contamination (C_d), we may now define the potential ecological risk index (RI) as the sum of the risk factors, i.e.:

$$RI = \sum_{i=1}^{8} E_r^i = \sum_{i=1}^{8} T_r^i \cdot C_f^i$$

The following terminology may be used to describe the RI-values:

$RI < 150$ => low ecological risk for the lake/basin;

$150 \le RI < 300$ => moderate ecological risk for the lake/basin;

$300 \le RI < 600$ => considerable ecological risk for the lake/basin;

$RI \ge 600$ => very high ecological risk for the lake/basin.

problems in the data, they can be used to demonstrate the use of the potential ecological risk index.

Most of the lakes examined are influenced by mining activities and the presence of a mine or mines was the basis of the studies. Rice Lake was studied before and after the reactivation of San Antonio Gold Mine in Bisset, Manitoba. Trout and Cliff lakes were studied in association with Trout Lake Mine; however, most of the contamination is believed to be derived by aerial deposition from smelter stack emissions at Flin Flon, Manitoba. Both Cliff and Trout lakes are the source of Flin Flon's water supply. Snow and Herblet lakes, Manitoba were investigated in relation to a gold mine which was operational from 1949 to 1958. Snow Lake is utilized as a water supply by the Town of the same name. Gods Lake, Manitoba was investigated for potential pollution by leachate from an abandoned gold mine. Gods Lake Gold Mines Limited operated for eight years from 1935 to 1943. Francois Lake, B.C. was investigated in connection with an amendment of Endako Mines pollution control permit. Discharges were allowed from their new ultra pure plant to a new No. 3 tailings pond and evaluation of seepages from their No. 1 and No. 2 tailings ponds was required. Meg, Keg, Peg, Narrow and Great Slave lakes, N.W.T. were studied to determine the effects of waste discharges from Cominco Ltd.'s Con Mine operations in Yellowknife. The tailings pond effluent flows from Mey Lake to Keg Lake to Peg Lake and finally, to Great Slave Lake. Narrow Lake is not in the path of the tailings effluent and was considered a "control" lake. Kootenay Lake, B.C. has been influenced by many mines since 1890, the largest and lonyest producing mine being the Bluebell near Riondel. The lake was used as a site for dumping waste and disposal of tailings. Sun Bay (Contwoyto Lake), N.W.T. was sampled to evaluate the effects of tailing pond decants from the Echo Bay Mines, Lupin Gold Mine. The values used to determine the contamination factors for Sun Bay are from the post-decant period.

6.3.2 Sediment Data

The sediment data for the selected lakes are presented in Table 17. For this example, any missing values for the metals were set as equal to the natural background levels (C_n^i) previously defined (Section 6.2.1). The bioproduction index (BPI) required to define the sensitivity of the lakes to pollutants was determined from the mean total phosphorous values expressed as ug/L. The BPI was then estimated using a graph relating total phosphorous to BPI (Figure 5 in Hakanson 1980A). This was necessary as most of the sediment data did not include the ignition loss and nitrogen values needed to estimate the BPI from sediments. The mean total phosphorous values were generally lake-wide means but were not always annual means (i.e. often they were based on one sample date only). Thus the BPI-values determined must be considered as crude estimates.

6.3.3 Contamination Factors and Degree of Contamination

The contamination factor (C_f^i) for each substance in the selected lakes was calculated by dividing the mean concentration of the metal in surficial sediments by the preindustrial reference value for that metal (Table 18). To derive degree of contamination values comparable to Swedish data, an arbitrary contamination factor (C_f^i) of 1 was assigned for PCB and all eight contamination factors were summed (Table 18). The range of degree of contamination was considerable among the lakes selected. The lakes were then ranked according to the degree of contamination, the pollutants were ranked by the contamination factor, and these were, in turn, compared to 15 Swedish lakes (Table 19).

The synoposis of large amounts of data (Table 19) by a standard method utilizing well-defined terminology can be very valuable tool for decision-makers in aquatic pollution control. The lakes near the top of the list had the highest degree of contamination. For each

Table 17. Mean values for constituents of surficial sediments (usually 0-5 cm) from selected Canadian lakes (see Table 16 for data sources).

Lake	P_{H2O} (ug/L)	Est. BPI	As	Cd	Cr	Cu	Pb	Hg	Zn
			(ug/g dry substance)						
Rice	25	4.9	32.7	1.93	90”	610	33.7	1.42	202
Snow (Block B)	38	5.2	112.9	1.4	90*	58.1	35.0	0.07	10
Herb1 et (Block E)	78	6.0	273.4	1.0	90”	654	27	0.03	119
Trout	10	3.5	175	10	90*	318	266	0.40	1502
Cliff	14	3.9	107	17	90*	620	396	1.22	2922
Gods (Block E)	20	4.5	8.5	1.6	90*	37.9	20.6	0.30	83.4
Francois	6	2.8	20.4	0.6	22.8	31.8	9.0	0.25*	76
Meg	115	6.0	539	ND	33	477	11	0.132	112
Keg	220	6.0	349	ND	120	544	8	0.047	252
Peg	76	6.0	76	ND	89	106	8	0.080	185
Narrow	63	5.9	22	ND	38	39	8	0.037	82
Great Slave	25	4.9	12	ND	130	172	14	0.053	199
Kootenay	15	3.9	193	2.92	22	57	567	0.25*	691
Sun Bay	<20	3.9	7.17	0.13	37.0	23.9	3.5	0.012	47.2

* Hypothetical values set to preindustrial reference levels.

ND = Not Detected.

Table 18. Contamination factors (C_f^i) and values illustrating the degree of sediment contamination (Cd) in 14 Canadian lakes. **All figures** marked with an * are hypothetical.

Lake	C_f^i								
	As	Cd	Cr	Cu	Pb	Hg	Zn	PC6	Cd $= \sum_{i=1}^{8} C_f^i$
Rice	**2.2**	1.9	1.0*	12.2	0.5	5.7	1.2	1.0”	25.6
Snow	**7.5**	1.4	1.0”	1.2	0.5	0.3	0.6	1.0*	13.5
Herblet	**18.2**	1.0	1.0”	13.1	0.4	0.1	0.7	1.0”	35.5
Trout	**11.7**	10.0	1.0*	6.4	3.8	1.6	8.6	1.0*	44.0
Cliff	**7.1**	17.0	1.0”	12.4	5.7	4.9	16.7	1.0*	65.8
Gods	**0.6**	1.6	1.0	0.8	0.3	1.2	0.5	1.0*	6.9
Francois	**1.4**	0.6	0.3	0.6	0.1	1.0*	0.4	1.0*	5.4
Meg	**35.9**	0.0	0.4	9.5	0.2	0.5	0.6	1.0*	48.2
Keg	**23.3**	0.0	1.3	10.9	0.1	0.2	1.4	1.0”	38.2
Peg	**5.1**	0.0	1.0	2.1	0.1	0.3	1.1	1.0”	10.7
Narrow	1.5	0.0	0.4	0.8	0.1	0.2	0.5	1.0”	4.4
Great Slave	0.8	0.0	1.4	3.4	0.2	0.2	1.1	1.0*	8.2
Kootenay	12.9	2.9	0.2	1.1	8.1	1.0*	3.6	1.0*	30.8
Sun Bay	0.5	0.1	0.4	0.5	0.1	0.1	0.3	1.0*	3.0

Table 19. Ranking of sediment contamination for the selected Canadian and Swedish (Hakanson 1980A) lakes according to the degree of contamination (Cd) value and the order of contamination factors (C_f^i). All values marked * are hypothetical.

Degree of Contamination (C_d)	Lake	Cd	Contamination Factors (C_f^i): Very High $C_f^i \geq 6$	Considerable $3 \leq C_f^i < 6$	Moderate $1 \leq C_f^i < 3$	Low $C_f < 1$
Very High $Cd \geq 32$	Stora Aspen	75.9	Cr > Pb > Zn > As*	PCB* > Cd	Cu > Hg	
	Cliff	65.8	Cd > Zn > Cu > As	Pb > Hg	Cr* > PCB*	
	Norra Barken	49.8	Zn > Pb > Cd > PCB*	AS* > Hg	cu	Cr
	Meg	48.2	As >> Cu		PCB*	Zn > Hg > Cr > Pb > Cd
	Vasman	47.2	Pb > Hg	Zn > Cd > As* > PCB*	cu	Cr
	Trout	44.0	As > Cd > Zn > Cu	Pb	Hg > Cr* = PCB	
	Kea	38.2	As >> Cu	-	Zn > Cr* > PCB*	Hg > Pb > Cd
	Herblet	35.5	As > Cu		Cd = Cr* = PCB*	Zn > Pb > Hg
	Ovre Hillen	34.6	Hg > Zn	PCB* > Cd > As* > Pb	cu	Cr
	Ostersjon	33.9	Cr > Zn	As* > PCB	Cu > Hg > Pb > Cd	
	Amanningen	32.1	Zn	Pb > Cr > PCB* > As*	Cd > Hg > Cu	
Considerable $16 \leq Cd < 32$	Kootenay	30.8	As > Pb	Zn	Cd > Cu > Hg* = PCB*	Cr
	Rice	25.7	Cu	Hg	As > Cd > Zn > Cr* = PCB*	Pb
	Freden	24.8	Zn	Cr > PCB* > Cd > As*	Hg > Cu > Pb	
	Blacken	18.6	-	PCB* > Zn	Cr > As* > Cu > Cd > Hg > Pb	
	Varmlandssjon	18.4	Hg		Zn > PCB* > As* > Cd > Pb	Cr = Cu
	Malaren	17.2	-	PCB*	Zn > Hg = As* = Cu > Pb > Cr > Cd	-
Moderate $8 \leq Cd < 16$	Vanern	15.9	-	Hg	Zn = PCB* = As* > Cd = Pb	Cr = Cu
	Hjalmaren	15.1	-	PCB* > As*	Cd* > Pb > Hg* > Cu	Zn > Cr
	Vattern	14.6	-	PCB*	Zn > Pb > As* > Cd	Hg > Cu > Cr
	Haggen	14.0	-	PCB = As*	Cd > Pb > Zn	Hg > Cu > Cr
	Snow	13.5	As		Cd > Cu > Cr* = PCB*	Zn > Pb > Hg
	Bysjon	10.9	-		PCB* = As* > Zn > Hg = Cd	Pb > Cu > Cr
	Peg	10.7	-	As	Cu > Zn > Cr = PCB*	Hg > Pb > Cd
	Great Slave	8.2	-	cu	cr > Zn > PCB*	As > Pb = Hg > Cd
Low $Cd < 8$	Gods	6.9	-		Cd > Hg > Cr = PCB*	Cu > As > Zn > Pb
	Francois	5.4	-		As > Hg* = PCB*	Cd = Cu > Zn > Cr
	Narrow	4.4	-		As > PCB*	Cu > Zn > Cr > Zn > Pb = Hg = Cd
	Sun Bay	3.0	-		PCB*	As = Cu > Cr > Zn > Pb = Hg = Cd

lake, the contaminants were ranked from highest (left) to lowest (right) contamination factor. Thus, if an environmental manager was reviewingan application to discharge pollutants into a lake that was catalogued in the list, he would have some rational basis for a response and subsequently, development of a discharge permit. Lakes with higher degrees of contamination could demand more stringent effluent controls and possibly no discharge of certain pollutants which already show a high contamination factor.

The degree of contamination is a useful tool which is based on few assumptions. It does not, however, account for the toxicity of the pollutant or the sensitivity of the water body. The toxicity and lake sensitivity would have to be taken into account by the manager when using such a table. In order to standardize the method whereby these two factors are taken into account, Hakanson (1980A) made numerous assumptions about toxicity, its expression and the sensitivity of the waterbody to produce risk factors (Eri) for each substance and a potential ecological risk index (RI) for water bodies. While risk indices are valuable diagnostic tools, they should not be used without due regard to the assumptions used in their derivation. Thus, it is beneficial initially to produce tables of contamination factors and degree of contamination for the lakes of a province or region which can then be examined by regional environmental managers aware of the special problems and sensitivities within their region.

From their high degree of contamination (Table 19) one would expect elevated metals or other problems to be evident in the biota of Cliff, Meg, Trout, Keg and Herblet lakes. This was confirmed by the conclusions in the respective reports. Cliff and Trout lakes had elevated levels of Cu, Zn and As in fish, with levels in Cliff Lake fish being higher. Meg and Keg Lakes both had reduced invertebrate numbers and elevated As, Pb, Zn and other metals in the water. Herblet Lake had elevated levels of As in fish and water.

One would expect lower levels of metals and associated problems to be evident in Kootenay and Rice lakes which both displayed a considerable degree of contamination. Elevated levels of Cu and As were observed in fish from Rice Lake. Concentrations of metals were low in Kootenay Lake fish; this could have been due to a low biologitally-active component of the metals, the localized nature of the metal contamination and the age of the contamination. Lakes with low to moderate degrees of contamination (Snow, Peg, Great Slave, Gods, Francois, Narrow and Sun Bay) would be expected to display the least or no problems with metal contamination in biota. The conclusions of the respective reports confirmed this assumption. Snow and Great Slave lakes showed low levels of metals in fish tissue. Both Peg and Great Slave lakes had an abundance of benthic macroinvertebrates and Narrows Lake was a "control" lake for the study. Both Gods Lake and Sun Bay showed low metal concentrations in fish tissue; however, some bioaccumulation was demonstrated in the macroinvertebrates in both waterbodies. The Francois Lake report was a data report with no conclusions drawn, but metal levels in fish tissue appeared to be low. For a discussion of the Swedish lakes see Hakanson (1980A).

The contamination character of a particular lake/basin may be described in a uniform, instructive and standardized way by using the contamination factor and the degree of contamination. This is useful in aquatic pollution control since it provides a basis for discussion and decisions by government administrators, resource developers and resource managers.

6.3.4 Risk Factors and a Potential Ecological Risk Index

Contamination factors and degree of contamination deal with the "easy points" of this approach and are primarily concerned with concentration and numbers of contaminants. They do not directly address the problem of ecological risk and effects. In an attempt to account

Table 20. Risk factors (E_r^i) and risk indices (RI) for the investigated lakes and for a reference lake (with $C_f^i = 1$ for all substances and BPI = 5.0). All values marked with an * are hypothetical and are only included to illustrate the principle.

Lake	E_r^i As	Cd	Cr	Cu	Pb	Hg	Zn	PCB	$RI = \sum_{i=1}^{8} E_r^i$
Rice	22	59	2"	62	2	232	1	39*	419
Snow	75	41	2*	6	2	11	1	42”	180
Herblet	182	27	2"	60	2	4	1	48*	326
Trout	117	359	2"	38	22	91	10	28*	668
Cliff	71	577	2*	70	32	250	19	31*	1054
Gods	6	51	2*	4	2	53	1	36”	154
Francois	14	24	1	4	1	71*	1	22*	138
Meg	359	0	1	44	1	18	1	48*	470
Keg	233	0	2	50	1	6	1	48*	340
Peg	51	0	2	10	1	11	1	48*	122
Narrow	15	0	1	4	1	5	1	47*	72
Great Slave	8	0	3	17	1	9	1	39*	78
Kootenay	129	99	1	6	46	51*	4	31*	367
Sun Bay	5	4	1	3	1	2	1	31*	47
Reference	10”	30"	2"	5*	5*	40”	1”	40”	133*

Table 21. The potential ecological risk factor (E_r^i) and risk indices (RI) of the investigated Canadian and Swedish (Hakanson 1980A) lakes. All figures marked with an * are hypothetical.

RISK INDEX	Lake	RI	Potential Ecological Risk Factor (E_r^c): Very High $E_r^i \geq 320$	High $160 \leq E_r^i < 320$	Considerable $80 \leq E_r^i < 160$	Moderate $40 \leq E_r^i < 80$	Low $E_r^i < 40$
Very High RI $\geq$ 600	Vasman	1201	Hg	Cd	PCB* > PB	As*	Cu > Zn > Cr
	Cliff	1054	Cd	Hg	-	As > Cu	Pb > PCB* > Zn > Cr*
	Norra Barken	813	-	PCB* > Cd	Hg	Pb > Cu	Zn > Cu > Cr
	Ovre Hillen	741	Hg	PCB*	Cd	Pb > As*	Pb > Cu > Zn > Cr
	Trout	668	Cd	-	As > Hg	-	Cu > PCB* > Pb > Zn > Cr*
	Varmlandssjon	652	Hg	-	-	Cd > PCB*	As* > Pb > Cu > Zn > Cr
	Stora Aspen	650	-	PCB*	Cd > As*	Hg > Pb > Cr	Zn > Cu
Considerable 300 $\leq$ RI $\leq$ 600	Vanern	511	Hg	-	-	Cd > PCB*	As* > Pb > Cu > Zn > Cr
	Meg	470	As	-	-	PCB* > Cu	Hg > Pb = Zn = Cr > Cd
	Rice	419	-	Hg	-	Cu > Cd	PCB* > As > Pb = Cr* > Zn
	Osterjon	415	-	-	PCB* > Hg	Cd > As*	Cr > Cu > Pb > Zn
	Hjalmaren	380	-	PCB*	-	Cd*	Hg* > As* > Pb > Cu > Cr > Zn
	Amanningen	370	-	PCB*	-	Cd > Hg > As*	Pb > Cr > Zn > Cu
	Kootenay	367	-	-	As > Cd	Hg* > Pb	PCB* > Cu > Zn > Cr
	Freden	365	-	PCB*	Cd	Hg	As* > Cr > Pb = Cu = Zn
	Keg	340	-	As	-	Cu > PCB*	Cr > Pb = Zn > Cd
	Malaren	326	-	-	PCB* > Hg	Cd	As* >Cu > Pb > Cr > Zn
	Herblet	326	-	As	-	Cu > PCB*	Cd > Hg > Pb > Cr* > Zn
	Blacken	305	-	PCB*	-	Cd > Hg	As* > Cu > Cr > Pb > Zn
Moderate 150 $\leq$ RI $\leq$ 300	Haggen	275	-	-	Cd	PCB* > Hg	As* > Pb > Cu > Zn > Cr
	Vattern	248	-	-	PCB*	Cd* > Hg	As* > Pb > Cu > Zn > Cr
	Bysjon	231	-	-	PCB*	Hg > Cd	As* > Pb > Cu = Zn > Cr
	Snow	180	-	-	-	As > PCB* > Cd	Hg > Cu > Pb = Cr* > Zn
	Gods	154	-	-	-	Hg > Cd	PCB* > As > Cu > Pb = Cr* > Zn
Low RI < 150	Francois	138	-	-	-	Hg*	Cd > PCB* > As > Cu > Cr = Pb = Zn
	Reference	133*	-	-	-	Hg* = PCB*	Cd* > As* > Pb* = Cu* > Cr* > Zn*
	Peg	122	-	-	-	As > PCB*	Hg > Cu > Cr > Pb = Zn > Cd
	Great Slave	78	-	-	-	-	PCB* > Cu > Hg > As > Cr > Pb = Zn
	Narrow	72	-	-	-	PCB*	As > Hg > Cu > Cr = Pb = Zn > Cd
	Sun Bay	47	-	-	-		PCB* > As > Cd > Cu > Hg > Pb = Zn = Cr

for the potential transport avenues of toxic substances to man, their threat to man, and their more complex threat to the aquatic ecosystem, risk factors (Er^i) and a potential risk index (RI) were developed as outlined in the preceding sections. The risk index approach expresses potential for a toxic response to substances (Tri) as a function of the natural abundance of the substance in the environment, the proportional representation of the substances in the sediments ("fingerprint") (Section 6.1.4) and the lake sensitivity to the substances as a function of bioproduction (Section 6.1.5).

The risk factors (Eri) for the selected lakes were calculated (Table 20) by multiplying the toxic response factor (Tr^i) by the contamination factor (C_f^i). The risk index (RI) is simply the sum of the eight risk factors for a given lake. The lakes were then ranked by the risk index and the contaminants were ranked by risk factors (Table 21). The risk index ranked the lakes in a slightly different order than did the degree of contamination (Table 19). Cliff and Trout lakes still headed the list among Canadian lakes due to high contamination and their oligotrophic nature. Meg, Keg and Herblet lakes were rated lower due to their highly eutrophic status which decreases their sensitivity to metal pollution. Kootenay and Rice lakes remained at the same level. The remaining lakes showed only slight shifts in the low to moderate categories depending on the primary contaminant (e.g. Gods Lake moved up because of the emphasis placed on Hg by the risk index). The ordering of the contaminants by risk factors, however, changed substantially from the previous order obtained by contamination factors (Table 19). Greater importance is given to PCB, Hg, Cd and As by the toxic-response factor (Tri) and, consequently, these four substances dominated the very high and high risk factor categories. Thus, greater priority is placed on contamination by these substances than by, say, Zn contamination.

To obtain a full appreciation of the factors upon which this sedimentological index is built, the folloiwng example may be useful:

(a) What is the significance of a risk factor of Eri = 50? This is the limit between low and moderate potential ecological effect.

Assume that the lake has a BPI-value of 5.0.

The result for the given substances are shown in Table 22.

The toxic factors in the second column should be interpreted as normative constants. The contamination factors in the third column are determined from the equation:

$$C_f^i = \frac{50}{Tr^i}$$

The preindustrial reference values for the sediments (C_n^i) should be considered as constants.

The mean superficial sediment contents required to obtain an Eri of 50 for the various substances (last column) have been determined from the equation:

$$C^i_{0-1} = C^i_f \cdot C^i_n$$

The last column illustrates that a sedirnent content of 8750 ppm for Zn, 2250 ppm for Cr, 500 ppm for Cu, 700 ppm for Pb, 1.67 ppm for Cd, 0.31 ppm for Hg or 0.013 ppm for PCB would produce a low to moderate ecological risk in lakes with moderate levels of bioproduction (BPI = 5.0).

Table 22. The relationship between the contamination factors (Cfi) and the C^i_{0-1}-values (i.e. the mean content in ppm of the substances in superficial sediments 0-1 cm from accumulation areas) for two given risk factors (Er^i) of 50 and 320 and a constant bioproduction index (BPI) of 5.0 (Hakanson 1980A).

Substance	Toxic Factor (St^i-value)	Contamination Factor (C_f^i-value)	Preindustrial Reference Level (C_n^i-value)	Mean Superficial Sediment Content (C^i_{0-1}-value)
Er^i = 50, BPI = 5.0				
PCB	40	1.25	0.01	0.013
Hg	40	1.25	0.25	0.31
Cd	30	1.67	1.0	1.67
As	10	5.0	15	75
Pb	5	10.0	70	700
cu	5	10.0	50	500
Cr	2	25.0	90	2250
Zn	1	50.0	175	8750
Eri = 320, BPI = 5.0				
PCB	40	8.0	0.01	0.08
Hg	40	8.0	0.25	2.0
Cd	30	10.7	1.0	10.7
As	10	32.0	15	480
Pb	5	64.0	70	4480
cu	5	64.0	50	3200
Cr	2	160	90	14400
Zn	1	320	175	56000

The crucial point is not that this analysis is an absolute representation of risk, but rather that the values obtained provide a framework for comparisons. The approach provides data in a quick, inexpensive and standardized manner. These data should be looked upon as reference figures - a discrepancy from the reference value may often be just as important and informative as a value which seems to agree with present knowledge.

(b) It is even more interesting to apply the same test to the limitation value between high and very high potential ecological risk (i.e. Eri = 320). The results are also given in Table 22.

For moderately bioproductive lakes, with a BPI-value of 5.0, a sediment contamination of Zn = 56,000 ppm, Cr = 17,700 ppm, Cu = 3,200 ppm, Pb = 7,780 ppm, As = 780 ppm, Cd = 10.7 ppm, Hg = 2.0 ppm or PCB = 0.08 ppm would yield the same ecological risk factor (Er^i = 320).

What do these examples say about the validity of the ecological risk factor and how we can check these interpretations? What is the consequence of a high ecological risk factor to biological systems, species or organisms? Existing information is insufficient to verify or dispute these interpretations. Some examples may illustrate this point. A value of 2 ppm in surficial sediments for Hg would produce a hiyh ecological risk factor and many eutrophic lakes with C^i_{0-1}-values for Hg in this range (e.g. lake Ovre Hillen where C^i_{0-1} = 2.04) are posted as unsuitable for fishing. If we take the value for Zn as another example, we would rarely expect to find C^i_{0-1}-values in the range of 60,000. Such sediment contents have been obtained in zinc-sand deposits outside zinc-mines. While the bottom fauna has been considerably altered and reduced in numbers, it is not clear whether this is due to high Zn contamination or is a consequence of other contaminants, such as Cd or Cu. At present it is not possible to answer such questions, and thus we have to accept the results for what they are (i.e. readily determined diagnostic values indicating that further investigations should be focused on elements and locations which show high or very high risk factors).

6.4 BENEFITS OF THE ECOLOGICAL RISK INDEX AND FUTURE DIRECTIONS

Data from 14 Canadian lakes have been assembled and analyzed in a stepwise progression to produce contamination factors for individual substances in individual waterbodies, values expressing the degree of contamination resulting from the influence of several substances in each waterbody, risk factors for each substance in each lake incorporating expressions of toxicity and receiving lake sensitivity and, finally, risk indices which sum the risk factors for several substances acting in each waterbody. The result has been a ranking of lakes in terms of the ecological risk associated with their current contamination problems and a ranking of the contaminant of concern within each waterbody. The difficulties with the Canadian data available for this exercise have been discussed and, for this reason, the actual values obtained by this analysis should not be used outside the context of this report. The real value in this analysis lies in illustrating the principal of the risk index, demonstrating how data gathered from contaminant studies may be practically applied to aquatic pollution control and revealing ways to improve our data collection programs so that the information gathered is more useful to resource managers.

A major benefit of the risk index analysis is that it shows the potential for using data from studies across the country to systematically document our empirical observations on the behaviour of contaminants in various aquatic systems. With comparable data from numerous sites we have the opportunity to compare waterbodies and relate observed contamination conditions to their respective contaminant discharge scenarios. By systematically building this empirical evidence, we have a much stronger basis for making future contaminant management decisions.

In addition, the risk index has the benefit of translating data into terms which are directly relevant to resource managers. It provides them with a comparative assessment of the status of waterbodies and highlights waterbodies and contaminants of concern. Such information may be useful, for example, in directing monitoring efforts or defining waste discharge criteria.

In this review of Canadian data, it was found that inconsistencies in sampling design and protocol presently limit the deyree to which the data can be used for comparative analysis. It is anticipated that some standardization of sampling methods could be readily implemented without sacrificing individual study objectives. Specific recommendations in this regard include sampling sediments in accumulation zones as determined by bathymetric data, analyzing a consistent core fraction (e.g. 0-1 cm) and incorporating measurements of recipient -'sensitivity (e.g. bioproduction as a function of sediment organic content).

A systematic effort to develop natural background or preindustrial levels (Cni) for given jurisdictional or physiographic areas in Canada would provide greater resolution for contaminant factors than the global levels used in this illustration. At the same time, a regional data base would help to place limited site-specific baseline studies in perspective. It is recoynized that a great deal of federal and provincial cooperation is required to implement any standardization of sampling methods, but such an effort is needed and would pay lony-term dividends.

Use of the .framework, and the risk index analysis in particular, opens up many avenues for further investigation. There are many interesting and important aspects that have been deliberately ignored in the development of the risk index to date. These include:

(a) The impact of pH, Eh, alkalinity and other water parameters on the toxicity of the substances.

(b) The impact of morphometry and hydrology on the sedimentological "fingerprint".

(c) Alternative expressions of recipient sensitivity (e.g. chlorophyll and algal volume).

(d) The relationship between the type of pollution, the species of metal and the toxicity (i.e. a measure of biological availability and/or toxicity).

(e) Alternative substances or groups of substances (e.g. oil, PAH, pesticides and others).

In a Canadian context, these issues should be addressed. Perhaps a risk index for metal pollution should be developed separately from a risk index for organic pollution. The substances included should be relevant to the particular physiographic or manayement region. As noted earlier the development of preindustrial levels ($C_n{}^i$) for individual physiographic areas may provide greater resolution, while the presently defined levels could be used on a national basis.

The risk index as illustrated applies specifically to lake systems and emphasizes a sedimentological approach using data from accumulation zones. Many contaminant discharges occur in rivers and other areas where accumulation sediments cannot be employed. Risk indices should be developed for these systems using alternative appropriate expressions of dose, recipient sensitivity and effect (e.g. concentrations in suspended sediments, concentrations/biomass for benthic biota, production/respiration ratios, etc.).

The overall objective of the framework, including the risk index, is to increase our understanding of the 'operative linkages between contaminant dose and environmental response; thus, a major thrust for further work is the incorporation of direct measurements of ecological risk (i.e. biological effect parameters in the risk index). Systematic development of empirical data for these relationships can in turn lead to development of diagnostic and prognostic models of environmental impact as discussed in Section 7.

7.0 THE APPLICATION OF DIAGNOSTIC AND PROGNOSTIC MODELS IN CANADA

7.0 THE APPLICATION OF DIAGNOSTIC AND PROGNOSTIC MODELS IN CANADA

7.1 INTRODUCTION

The successful application of the conceptual framework in Canada would generate a homogeneous data set for dose, recipient sensitivity and ecological effect terms for many lakes. From this data, set empirical relationships between dose and response could be developed and explored. These relationships could then be used to produce mathematical models. In some cases, the mathematical models produced would be intuitively obvious based on present ecological knowledye. Conversely, the empirical relationships might not be intuitively obvious and future research could be directed at the underlying causes of the relationship. The developed models could be used as diagnostic and prognostic tools in aquatic environmental assessment.

Using the framework approach, Hakanson (1980B) has developed a model for mercury contamination in fish. Subsequent sections of this report outline the derivation of this model and illustrate its potential application in aquatic pollution control. The mercury model is tested using available Canadian data. Finally, the general status of the Canadian data base for model development is reviewed and recommendations made for improvements.

7.2 A MODEL FOR MERCURY CONTAMINATION

A formula for mercury contamination in fish was deduced by Hakanson (19805) from a large, inhomogeneous data source and was tested with very positive results on an independent set of Swedish lakes. The formula describes the quantitative impact of pH, bioproduction and mercury contamination of the sediments on the mercury content of northern pike (*Esox lucius*). The formula is:

$$F(Hg) = \frac{4.8 \times \log (1 + Hg_{50}/200)}{(pH - 2) \times \log BPI}$$

where

F(Hg) = the content of methylmercury in 1 kg pike muscle tissue (as mg/kg wet weight);

Hg_{50} = the weighted mean mercury content of surface sediments (0-1 cm) in ng/g dry substance;

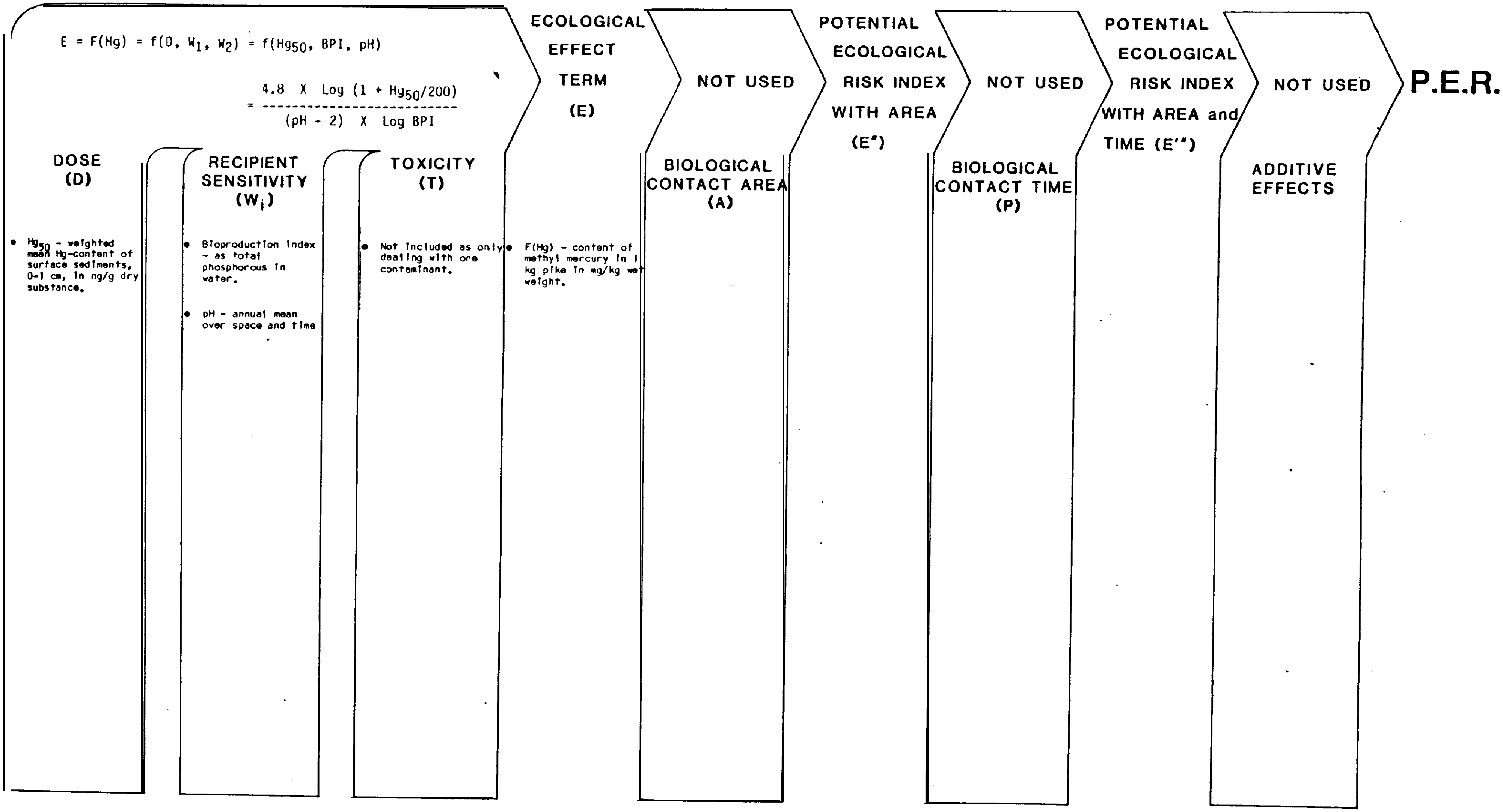
PH = the mean pH of the water system;

BPI = the bioproduction index.

This formula is primarily meant to be used as a diagnostic tool in practical work concerning aquatic pollution control. The model incorporates the most important aspects of the conceptual framework (Figure 3) by relating dose and recipient sensitivity to an ecological effect term (Fiyure 6).

The model incorporates assumptions which place certain demands on the data. The mean pH value should be determined on different occasions (months, years), at various sites and at different elevations in the water column. A rule of thumb for the minimum number of pH values would be no less than five measurements of which at least two represent different seasons. The level of bioproduction (BPI) should be expressed as the value derived from 0-1 cm sediments (Hakanson 19808); however, estimates of BPI may be determined with less accuracy by other means (e.g. total phosphorous concentrations in the water). Mercury contamination (Hg_{50} in ng/g ds) should be expressed as an estimate of the areal median mercury content based on at least five sediment samples (0-1 cm) which provide an even coverage of the investigated area. The formula should only be used

Figure 6. Schematic of the conceptual framework illustrating the components and terms used in developing a model for mercury contamination in fish.



for lakes or basins with accumulation bottoms. As discussed in previous sections, basins with transportational or erosional bottoms yield sediment data which cannot be adequately interpreted for this application. Thus, this dose parameter cannot be used in shallow lakes where the entire bottom is dominated by processes of transportation and resuspension.

7.3 POTENTIAL APPLICATIONS OF THE MERCURY MODEL

Quantitative predictions of environmental impact are a major, and as yet unobtained, objective of EIA. The mercury model makes significant progress towards that objective. The predicted value, mercury concentration in fish tissue, is directly relevant to management and regulatory concerns. With further work, the dose parameter may be directly linked to project inputs.

In its present form, the model may be usefully employed to weight ecological and economic issues in resource manayement problems. Two examples illustrate potential applications.

The first example concerns a lake which is receiving mercury and is being acidified by atmospheric input related to the use of fossil fuels. The mercury dose and reduced pH have caused elevation of mercury in fish tissue to levels that are unsuitable for human consumption. If the atmospheric inputs cannot be eliminated, what management options are available for this lake? The model shows that an increase in bioproduction or pH should cause a reduction in mercury levels in fish. The first option of increasing bioproductivity may not be desirable for aesthetic reasons, leaving the second option of increasing pH with lime.

How much lime would be needed? From the model, the required increase in pH can be calculated. With access to data on water chemistry, lake volume and retention time, it is fairly simple to calculate the amount of lime needed to get the desired increase in pH. A lime treatment would be a temporary cure with attendant long-term costs. In this and similar examples, the model may be used to optimize the economic resources, since it provides quantitative data on treatments and response.

A second example highlights an interesting and perhaps paradoxical consequence of sewage treatment. Before installation of a sewage treatment plant, a lake has an Hg_{50} in sediments of 600 ng/g, a bioproduction index of 4.4 and a pH of 7.0. This would represent a severely mercury polluted lake in a mesotrophic state with neutral pH. According to the model, the mercury in fish would be 0.90 mg/kg wet weight.

After installation of the treatment plant, the input of nutrients and mercury would be reduced. This would in turn result in lower bioproduction, lower pH due to reduction in bioproduction and a lower value of mercury in the sediments. Assuming that the lake after treatment would have a bioproduction index of 3.0, a pH of 6.6 and Hg_{50} of 380 ng/g, what would be the effects on the lake? The transparency would be better, the lake would look cleaner, the trophic state would be reduced and sediment concentrations of mercury would be reduced.

What about the mercury content in fish? Using the model with the assumed values for bioproduction, pH and Hg_{50}, a mercury content in fish tissue (F(Hg)) of 1.01 mg/kg wet weight is predicted. This value would be higher than the before treatment value and would be above the accepted level for human consumption in Sweden. If the assumptions are reasonable, the model suggests that the cost of a sewage treatment plant may be augmented by unanticipated ecological and social costs.

These examples are primarily meant to illustrate the potential benefits of such models and formulae when it comes to addressing complicated ecological and economic inter-relationships. If such quantitative models were developed, our capability for predicting impacts and optimizing resources would be greatly improved.

7.4 CANADIAN TEST DATA

Available Canadian data were reviewed with the objective of testiny the mercury model. The amount of Canadian data meeting all the requirements of the formula was small; however, using various assumptions, the Canadian data was analyzed to see if the model could predict values which were reasonably close to the empirical data for mercury content in northern pike. From our survey of Canadian data, seven lakes were found to have sediment, water and northern pike data. The seven lakes (Rice, Snow, Herblet, Trout, Cliff, Gods and Great Slave; see Table 16) met only a few of the requirements of the formula. Of these lakes only Snow, Herblet, Trout, Cliff, Gods and Great Slave lakes were deep enough to have accumulation bottoms. The sediments for these lakes were collected from the top 5 cm. Trout and Cliff lakes were sampled on an even-area basis while the rest were sampled close to the contaminating source. Snow, Herblet, Trout and Cliff lakes had water chemistry from one season only. The bioproduction value was estimated from a graph relatiny total phosphorous to BPI (Figure 1 in Hakanson 1980B). Bearing in mind the problems with the data, a predicted value for mercury in pike tissue was derived and then compared to the empirically measured value.

7.5 RESULTS AND DISCUSSION

The predicted values for mercury levels in northern pike compared reasonably well with the measured values for Snow, Herblet, Gods and Great Slave Lakes (Table 23). For Trout and Cliff lakes, measured values were substantially lower than the predicted values (Table 23). The reason for this anomaly is unclear.

Table 23. Data from six Canadian lakes used to test the mercury model (Hakanson 19808) and the model-calculated values of mercury in fish tissue.

Lake	Hg (ug/kg) Sediments, 0-5 cm	Total P_{H2O} ug/L	Estimated BPI	Mean PH	F(Hg) Predicted	F(Hg) Empiric
Snow (Blocks B&C)	1451	412	5.2	7.782	0.27	0.31
Herblet (Blocks E&F)	1551	652	5.8	7.60	0.27	0.36
Gods (Block C)	138[3]	20	4.5	7.91	0.28	0.3(
Trout	402	101	3.5	7.701	0.74	0.35
Cliff	1220	141	3.9	7.791	1.19	0.31
Great Slave (Near Con Mine)	53	25	4.8	7.23	0.14	0.1;

1 Adjusted to an average of maximum values from sites providing a more even area coverage.
2 From numerous stations for one sample date only.
3 Sample site 40 excluded (N=8).

Theoretically, for these latter two oligotophic lakes, the high mercury contamination of the sediments should have produced elevated levels of mercury in fish tissue. Was the model at fault or were factors, which were not incorporated in the model, acting in these two lakes? A similar discrepancy occurred for Lake Saxen in Sweden (Hakanson 1984A) and it was postulated that the lower observed mercury levels in the fish were a result of zinc, and possibly lead, antagonism. Lake Saxen was found to have high contamination factors for both zinc and lead. When the contamination factors for Trout and Cliff lakes were examined (Table 18; Section 6.2.3), the sediment contamination factors were found to be high for both lead and zinc. The sum of the sediment contamination factors for lead and zinc was 12.4 and 22.4 for Trout and Cliff lakes, respectively. The mean zinc concentrations in the water were reported as 0.30 and 0.39 mg/L for Trout and Cliff lakes, respectively (Wilson 1984). In the case of these two lakes and Lake Saxen, the concurrent contamination by zinc may have acted antagonistically to the mercury pollution, thus providing one possible explanation for the model's failure to predict the measured values.

From this limited set of data for Canadian lakes and results of tests on Swedish lakes, the mercury model showed potential as a tool for predicting mercury levels in pike muscle tissue. The formula could be transformed from a diagnostic or descriptive model into a prognostic model if the Hg_{50}-value is replaced by a dose factor expressing mercury loading to the water system, and into a simulation model if various plausible mercury doses are tested. Such a model is greatly needed in practical pollution control. It could be used to examine such questions as: What potential ecological effect could be expected if X kg mercury were discharged to Lake A? Would Lake A be unsuitable for use as a domestic, sport or commercial fishery? What environmental factors are most important? If adequately tested quantitative models were available for the most important groups of toxic substances, then it would be possible to establish normal values and hence to quantify any divergence from the normal.

7.6 THE STATUS OF CANADIAN DATA FOR USE IN MODEL DEVELOPMENT

During the course of this study numerous Canadian government agencies (Appendix C) were contacted regarding availability of data on metal concentrations in lake water, sediments, invertebrates, and fish. No attempt was made to examine the so-called "grey" literature including unpublished impact assessment documentation and reports by other consultants.

This informal survey confirmed that standard data collection practices do not generally exist across Canada, although there is some standardization within provinces and agencies. Most of the persons contacted were very helpful and interested in the possibility of applying their data to the development of ecological risk and contamination models. As a result of our inquiries, numerous final reports, data reports, and raw data sheets were forwarded to us. It also became apparent that large amounts of data would not be comparable unless transformations were performed due to differences in data collection practices. As well, additional background information was required to enable accurate interpretation and appropriate normalization. Often, within data sets, values critical to the model development were missing.

One large standardized data set which showed considerable potential for application to model development was the water-sediment-fish data set collected by the B.C. Ministry of Environment, Water Manayement Branch. This computerized data set contained data for approximately 100 lakes for which there were approximately 40 with fish data. The data were collected using standardized collection and analysis techniques. While the data set was deficient for some parameters, a complete matrix of data including sensitivity and effect terms could be produced with a minimal effort. The number of relevant data sets for B.C. lakes also could be increased by a fish sampling and analysis program for the lakes currently missing fish data. Various other suitable individual data sets from other lakes in B.C. and across Canada could be transformed where necessary and incorporated into the matrix. From this data matrix useful risk models could be developed for various metal contaminants and several species of fish. The models could then be empirically tested by various agencies across Canada and further refined.

The data sets which have been collected for this study could also be applied to develop natural background levels of metals in sediments and fish for various regions in Canada. This would be a beneficial first step in the development and use of contamination factors in EIA's and aquatic pollution control programs (Sections 5 and 6).

7.7 RECOMMENDATIONS FOR IMPROVING THE DATA BASE

Previous sections have discussed deficiencies in available Canadian data with respect to developing quantitative aquatic contamination models. What, then, would constitute an ideal data base for the development of such useful quantitative formulae?

First, relevant data are needed from a representative sample of Canadian lakes. A representative sample might comprise 40 lakes for which the following environmental measurements have been determined: pH, alkalinity, major constituents, level of bioproduction (P/R, BPI, chlorophyll, etc.), transparency, water retention time, morphometry (volume, area, mean depth, maximum depth, etc.), areas of accumulation/erosion/transportation and area of sediment contamination. All these parameters/variables can be readily determined by existing methods.

It is not practical to study every possible toxic substance. Initially it may be more appropriate to select a group of representative substances, for example Hg, Cd, Pb, Cu, Cr, Ni, Zn, As, Al, PCB, DDT and PAH. A contaminant budget (input, output, sedimentation) should be determined to permit quantitative dose calculations. Data on the receiving water discharge (Q) should also be obtained.

For these substances, the following effect terms could be tested:

- concentration in aquatic plants (e.g. Fontinalis);
- benthic invertebrate community composition;
- concentrations in clam or snail shells; and
- concentrations in liver or tissue of appropriate fish species for given regions.

This strategy would yield a matrix of data, where the dose and sensitivity parameters could be correlated with the effect terms to identify major relationships and reduce the residual term (or the unaccountable remainder). The residual terms should be established by parallel empirical data sets. The results would, hopefully, yield useful risk models for the given substances and a better knowledge of the major causal relationships. If such empirically tested models were available, they would represent a major advance in our impact prediction and resource management capabilities compared to the somewhat chaotic situation of today.

8.0 CONCLUSIONS AND FUTURE DIRECTIONS

8.0 CONCLUSIONS AND FUTURE DIRECTIONS

8.1 INTRODUCTION

As stated at the outset of this report, the framework for aquatic contamination impact assessment represents one possible avenue for improving our impact assessment capabilities. As it stands the framework provides numerous practical and scientific benefits to the practice of EIA; however, parts of the framework, which are presently incorporated conceptually, require further development to enable functional integration into the framework. The framework is presented as a starting point to organize thinking and objectives. As more data becomes available and as elements of the framework are explored and developed, the framework will change and expand to reflect our growing knowledge. This section reviews the benefits of the framework approach discussed in previous sections, discusses potential for immediate application of the framework and identifies areas for further study.

8.2 BENEFITS OFFERED BY THE FRAMEWORK APPROACH

Previous sections have identified various benefits provided by the framework in enhancing the applicability of aquatic contaminant studies to impact assessment and in improving the scientific basis for contaminant studies. These benefits are reviewed briefly below.

1. The framework breaks the impact assessment problem down into several key information components (dose, sensitivity, toxicity, effect, contact area, contact time, additive effects). As such, it is a practical tool which stimulates investigators to systematically address and rationalize these important elements in their study design.

2. The framework assists in the definition of temporal and spatial dimensions for impact assessment study design.

3. The framework orients study design towards the practical objectives of impact prediction and resource management, thus increasing the chances that relevant information will be collected.

4. The simple format of the framework and its orientation towards objectives enhances communication amongst the study participants.

5. The framework, with its emphasis on the relationship between dose and response, encourages the integration of project design (engineering) studies with environmental studies, thus reducing the risk of irrelevant investigation in both areas arisiny from inappropriate assumptions.

6. Impact prediction is a major objective of impact studies. The framework provides the necessary building blocks for the formulation of quantitative impact prediction.

7. The framework approach strives for optimum representation of the possible ecological effect field by calling for effect terms which are appropriate integrators of effects in relation to contact time characteristics. In addition, the framework approach stresses selection of effect terms which provide linkages to the socially valued ecosystem components and which at the same time represent the weakest or most sensitive link in the contaminant pathway.

8. By orientiny studies towards practical objectives and focusing investiyations on information rich parameters, the framework can improve the cost effectiveness of impact studies.

9. The framework provides a number of practical tools applicable to aquatic contaminant assessment and aquatic resource manayement (contamination factors, risk indices, prognostic models). Use of these tools encourages sampling by standardized methods to generate comparable data. The availability of homogeneous data increases the usefulness of the data for comparative analysis and builds up a cumulative information base on contaminant effects in different water bodies.

10. The framework structure directs aquatic contaminant study emphasis away from the "end of the pipe" and the toxicology labs, and towards the receiving environment. Information on effects in the receiving environment provides a much stronger basis for management decisions.

11. The framework approach bases aquatic contamination impact studies on a thorough understanding of the physical dynamics of the receiving water body. This is essential to assigning appropriate spatial and temporal dimensions to aquatic impact studies.

12. The framework is not a rigid format for study design, but is a skeleton on which to organize study desiyn. It allows flexibility in the selection of specific parameters, but encourages thorouyh rationalization of that selection. The framework invites elaboration and refinement as knowledge of aquatic contaminant impacts grows.

8.3 APPLICATION OF THE FRAMEWORK APPROACH IN CANADA

Numerous studies of aquatic contaminant impacts are being carried out by industry and governments at all levels in response to formal EIA requirements, licence and permit requirements, monitoring proyrarn requirements and specific research objectives. Despite the large body of data that is constantly being generated, our understanding of aquatic contaminant impacts and our capability to manage them effectively is advancing relatively slowly. We submit that this situation might improve significantly if there was some impetus among aquatic contaminants investigators to communicate and coordinate study objectives, using a common frame of reference such as the conceptual framework. Realistically, widespread adoption of a common framework for analyzing contaminant problems and designing study methods could best be implemented through a government agency. Trial use of the framework by an existiny resource management agency (e.g. the Technical Advisory Committees or the Environmental Advisory Committee of the N.W.T. Water Board) or a specially constituted workshop of agency representatives in order to design an aquatic contaminant study and monitoring proyram would demonstrate the feasibility of implementing the framework approach. Feedback from such a trial application might, in turn, be used to refine the framework to improve its effectiveness as a communications tool and technical aid.

General applications of the framework approach to work carried out by agencies responsible for regulation, management and research related to aquatic contaminants include:

- Improved design of contaminant studies required by EIA, water licences, monitoring programs, etc.

- Standardization of sampling methods to generate data which is comparable between studies.

- Accumulation of information on contaminant behaviour in receiving environments through generation of contamination factors.

- Comparative assessment of lake contamination problems using risk indices.

- Development of prognostic models to facilitate impact prediction, determine mitigation requirements and permit analysis of costs and benefits related to various contaminant management options.

8.4 RECOMMENDATIONS FOR FURTHER STUDY

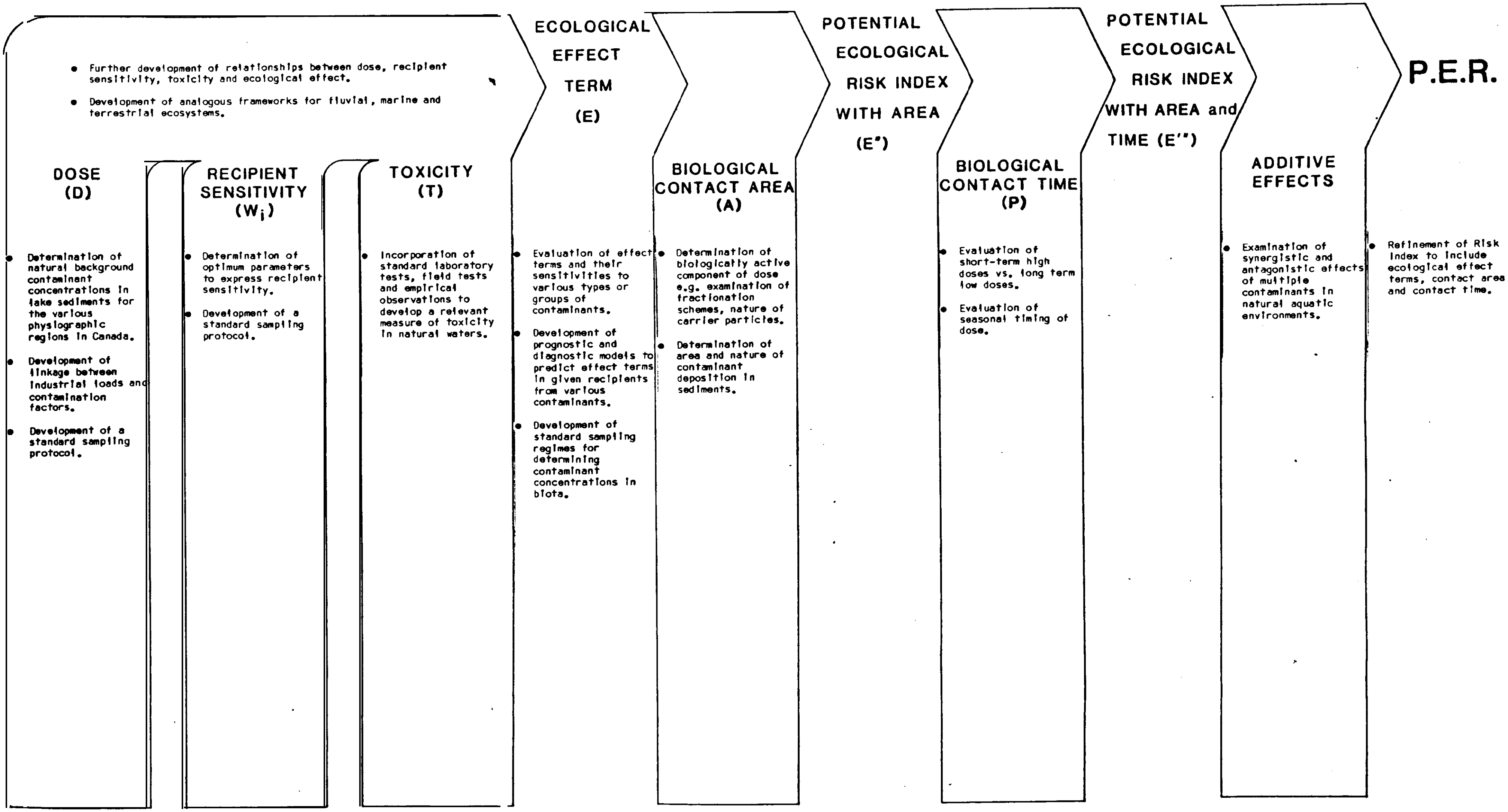
The framework, at its present level of development, offers many benefits to the study of aquatic contaminants. It also highlights current deficiencies in our understanding and information base related to aquatic contaminant impacts. The areas for further study (Figure 7) have been discussed in previous sections and are reviewed briefly below.

8.4.1 Further Development of Framework Linkages

- To date, information analysis using the framework notation has concentrated on integrating effect terms, dose, toxicity factors and environmental sensitivity factors. A major area of future development lies in the integration of contact area and contact time, vital parameters in any ecological study. Part of the problem lies in generating definitive information to truly define contact area, both physically and in terms of biological availability. Characterization of the physical dynamics of receiving environments and analyses to determine the fractionation characteristics of contaminant doses will aid development of these linkages.

- Prognostic and diagnostic models can be very useful tools in impact assessment and environmental management. A model to predict mercury levels in pike has been developed using Swedish data. Development of similar models for various contaminants and

Figure 7. Schematic of the conceptual framework illustrating some areas for recommended further research.



various species would be a very valuable exercise in Canada. This would require a considerable effort in researching unpublished data sources and in organizing, normalizing and analysing the assembled data. Implementation of standard sample collecting methods for contaminant studies also would facilitate this work greatly.

- Effect terms used in this model to date (e.g. metal levels in fish) reflect specific management concerns related to specific contaminants for which we have some understanding of mediating sensitivity factors. Much work can be done to explore the use of other effect terms in the context of metals and other contaminant problems. The possibilities are unlimited for exploring the application of this framework to assessment of many aquatic pollution problems.

8.4.2 Data Requirements

The ability to explore ecotoxicological linkages within the framework depends on the availability of comparable, relevant data. Data deficiencies which presently impede development of linkages include:

- contaminant dose or loading data in conjunction with effect term data;

- fractionation data as a means of defining contact area and time;

- data on contact area as determined by physical dynamics in aquatic systems; and

- consistent data for sensitivity parameters (e.g. measures of bioproduction).

Including these data requirements in government monitoring networks and permit sampling programs would greatly enhance the usefulness of the data base.

8.4.3 Sampling Protocol

A persistent problem in aquatic contamination studies is inconsistencies in the way samples are collected and analyzed. A valuable tool to enhance the use and transferability of information gathered in contaminant studies would be the development and implementation of standard data collection methods for measuring dose, sensitivity, toxicity and ecological effects in receiving waters.

8.4.4 Applying the Framework to Canadian Aquatic Environments and Concerns

Some of the assumptions used in deriving contamination factors and assigning risk factors to various contaminants are based on arbitrary assumptions or global generalizations concerning pre-industrial contaminant levels and their natural abundance. More resolution could be achieved in these analyses for site specific management applications in Canada if some systematic attempt was made to define preindustrial concentrations in physiographic and/or political/jurisdictional regions within Canada. Specifically, baseline information should be collected for water, sedirnent, clams, fish and other effect terms. Risk factors assigned to contaminants should be based on experience in Canada of problems associated with various contaminants. Based on these experiences, priority could be given to the systematic development of baseline data for problem elements.

8.4.5 Development of Analogous Frameworks for Other Receiving Environments

Lakes represent only one type of physical regime which is subject to contaminant discharges. The components of the framework are generic and it could just as validly be applied to other regimes (fluvial, marine, terrestrial). A major area for future work consists of developing expressions for dose, recipient sensitivity and effect which are appropriate to these other physical and biotic regimes.

8.5 CONCLUSIONS

This report has presented a case for adoption of the framework for aquatic contaminant assessment in the design and execution of aquatic impact studies. The study objectives (Section 3) outlined several criteria to improve EIA and the scientific validity of assessment studies. The success of the framework approach in addressing these criteria is discussed below.

In terms of enhancing scientific validity, the framework provides a checklist of basic components and relationships which govern contaminant impacts. BY selecting specific parameters and relationships for measurement, the investigator is forced to rationalize his choice and specify how factors such as time, space and additive effects will be addressed in his study design. By documenting this rationale, a clear trail of scientific logic (hypothesis formulation) is laid which may ultimately be confirmed, rejected or modified as new information is generated. The framework incorporates some well developed tools for quantification of effects, and the potential for refining and expanding these, as well as developing others, is unlimited. In addition, by encouraging hypothesis formulation and quantitative prediction, the framework establishes a firm basis for effects monitoring programs and the systematic accumulation of knowledge gained through aquatic contaminant impact studies.

From the standpoint of the EIA process, a significant benefit of the framework is the full integration of contaminant dose and ecological response terms. This perspective assists a clear definition of overall study objectives and ensures integration of study components that are often segregated into environmental and project planning/ engineering compartments. Further, the framework, with its emphasis on selection of a limited number of representative effect terms, assists in focusing study efforts on significant impacts and effect terms which are valued socially (i.e. valued ecosystem components). This approach, together with the requirement for comparable data generated by accepted, standard techniques, also enhances the cost-effectiveness of studies, a major benefit to project proponents/study sponsors. Finally, the quantitative analysis tools which assign relative dimensions to ecological risk associated with a number of impact problems or which can generate quantitative predictions for effect terms such as metal concentrations in fish tissues, can provide useful information to resource managers. These types of information can help managers to allocate study resources appropriately or provide a concrete rationale for establishing discharge standards.

The efficiency of the framework in promoting communication and consensus among EIA participants has been demonstrated in Sweden; however, this benefit remains to be tested in Canada. In support of its potential is Beanlands and Duinker's (1983) observation with respect to EIA that,

> "significant scientific improvements will depend on the early adoption of appropriate conceptual frameworks and technical standards to guide the required studies, as well as a recognition of the overriding constraint of time in the design of the assessment program".

We feel that the conceptual framework provides a useful starting point for improving aquatic contaminant impact assessment skills. Our search for Canadian data applicable to the framework analysis generated considerable interest and willingness to assist among those contacted. Many agencies have data for various parameters within the framework which they would like to see applied to the development of tools for aquatic pollution control.

Finally, it is important to re-emphasize Beanlands and Duinker's (1983) conclusion that the impetus to raise the scientific standards of impact assessment as a whole must come from administering agencies. It is hoped that this review will stimulate interest and enthusiasm among readers to experiment with the framework approach and to embark upon some of the future directions recommended in this report.

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APPENDIX A

Appendix A. Chemical analysis of water from Contwoyto Lake, N.W.T. for November 1985.

Location: Inner Sun Bay, south of Narrows
Sample No.: EPS Site 3, mid-depth
Date/Time: November 22, 1985, 13:10

		Lab Sample No.		
		1A	1B	1C
pH (pH units)		6.82	6.94	6.66
Conductivity (µS/cm)		61.5	64.4	60.7
Total Suspended Solids		L 1.0	L 1.0	L 1.0
Total Dissolved Solids		52.	51.	50.
Bicarbonate Alkalinity	HC03	9.82	9.82	9.82
Carbondte Alkalinity	co3	Nil	Nil	Nil
Ortho Phosphorus	P	0.003	0.002	0.005
Total Dissolved Phosphorus	P	L 0.2	L 0.2	L 0.2
Total Phosphorus	P	L 0.02	L 0.2	0.025
Nitrate and Nitrite Nitrogen	N	0.21	0.20	0.24
Ammonia Nitrogen	N	0.94	0.92	U.84
Organic Nitrogen	N	1.76	1.32	1.41
Total Kjeldahl Nitroyen	N	2.70	2.24	2.25
Total Organic Carbon	C	3.4	2.9	3.6
Total Cyanide	CN	L 0.01	L 0.01	L 0.01
Total Metals				
Aluminum	Al	L 0.15	L 0.15	L 0.15
Antimony	Sb	L 0.15	L 0.15	L 0.15
Arsenic	*As	0.024	0.022	0.018
Barium	Ba	0.004	0.004	0.005
Beryllium	Be	L 0.005	L 0.005	L 0.005
Bismuth	Bi	L 0.5	L 0.5	L 0.5
Boron	B	L 0.01	L 0.01	L 0.01
Cadmium	*Cd	L 0.001	L 0.001	L 0.001
Calcium	Ca	3.49	3.54	3.48
Chromium	*Cr	L 0.001	L 0.001	L 0.001
Cobalt	Co	L 0.02	L 0.02	L 0.02
Copper	*Cu	0.002	0.002	0.002
Iron	Fe	0.10	0.10	0.10
Lead	*Pb	L 0.001	L 0.001	L 0.001
Magnesium	Mg	0.56	0.56	0.56

Appendix A. CONT'D

Location: Inner Sun Bay, south of Narrows (cont'd)
Sample No.: ESP Site 3 mid-depth
Date/Time: November 22, 1985, 13:10

		Lab Sample No.		
		1A	1B	1C
Manganese	Mn	0.007	0.006	0.006
Mercury	Hg	L 0.00005	L 0.00005	L 0.00005
Molybdenum	Mo	L 0.04	L 0.04	L 0.04
Nickel	*Ni	L 0.005	L 0.005	L 0.005
Phosphorus	PO4	L 0.4	L 0.4	L 0.4
Potassium	K	0.72	0.72	0.72
Silicon	SiO_2	0.54	0.57	0.56
Silver	Ag	L 0.03	L 0.03	L 0.03
Sodium	Na	4.45	4.43	4.34
Strontium	Sr	L 0.001	L 0.001	L 0.001
Tin	Sn	L 0.03	L 0.03	L 0.03
Titanium	Ti	L 0.006	L 0.006	L 0.006
Vanadium	V	L 0.01	L 0.01	L 0.01
Zinc	*Zn	L 0.010	L 0.010	L 0.010

All results expressed as mg/L unless noted otherwise.

* Hydride Generation or Direct AA

L = Less Than

Appendix A. CONT'D

Location: Outer Sun Bay, north of Narrows
Sample No.: EPS Site 6, 1 m below surface
Date/Time: November 24, 1985, 11:45

		Lab Sample No.		
		2A	2B	2C
pH (pH units)		6.34	6.39	6.34
Conductivity (@/cm)		11.6	11.6	11.6
Total Suspended Solids		L 1.0	L 1.0	L 1.0
Total Dissolved Solids		10.	10.	10.
Bicarbonate Alkalinity	HC03	4.91	4.91	4.91
Carbonate Alkalinity	co3	Nil	Nil	Nil
Ortho Phosphorus	P	0.018	0.010	0.018
Total Dissolved Phosphorus	P	L 0.02	L 0.02	L 0.02
Total Phosphorus	P	L 0.02	L 0.028	L 0.02
Nitrate and Nitrite Nitrogen	N	L 0.010	L 0.010	L 0.010
Ammonia Nitrogen	N	0.075	0.033	0.033
Organic Nitrogen	N	0.32	0.52	0.33
Total Kjeldahl Nitrogen	N	0.39	0.55	0.36
Total Organic Carbon	C	1.5	1.3	1.4
Total Cyanide	CN	L 0.01	L 0.01	L 0.01
Total Metals				
Aluminum	Al	L 0.15	L 0.15	L 0.15
Antimony	Sb	L 0.15	L 0.15	L 0.15
Arsenic	"As	L 0.001	L 0.001	L 0.001
Barium	Ba	0.003	0.002	0.002
Beryllium	Be	L 0.003	L 0.003	L 0.003
Bismuth	Bi	L 0.5	L 0.5	L 0.5
Boron	B	L 0.01	L 0.01	L 0.01
Cadmium	*Cd	L 0.001	L 0.001	L 0.001
Calcium	Ca	0.78	0.77	0.77
Chromium	*Cr	L 0.001	L 0.001	L 0.001
Cobalt	co	L 0.02	L 0.02	L 0.02
Copper	*cu	L 0.001	L 0.001	L 0.001
Iron	Fe	L 0.03	L 0.03	L 0.03
Lead	*Pb	L 0.001	L 0.001	L 0.001
Magnesium	Mg	0.36	0.36	0.36

Appendix A. CONT'D

Location: Outer Sun Bay, north of Narrows (cont'd)
Sample No.: EPS Site 6, 1 m below surface
Date/Time: November 24, 1985, 11:45

		Lab Sample No.		
		2A	2B	2C
Manganese	Mn	L 0.003	L 0.003	L 0.003
Mercury	Hg	L 0.00005	L 0.00005	L 0.00005
Molybdenum	Mo	L 0.04	L 0.04	L 0.04
Nickel	*Ni	L 0.005	L 0.005	L 0.005
Phosphorus	PO4	L 0.4	L 0.4	L 0.4
Potassium	K	0.38	0.38	0.38
Silicon	SiO_2	0.13	0.096	L 0.08
Silver	Ag	L 0.03	L 0.03	L 0.03
Sodium	Na	0.43	0.41	0.40
Strontium	Sr	0.005	0.005	0.005
Tin	Sn	L 0.03	L 0.03	L 0.03
Titanium	Ti	L 0.006	L 0.006	L 0.006
Vanadium	V	L 0.01	L 0.01	L 0.01
Zinc	*Zn	L 0.010	L 0.010	L 0.010

All results expressed in mg/L unless noted otherwise.

* Hydride Generation or Direct AA

L = Less Than

Appendix A. CONT'D

Location: Outer Sun Bay, north of Narrows
Sample No.: EPS Site 6, 7 m depth
Date/Time: November 24, 1985, 11:45

		Lab. Sample No.		
		3A	3B	3c
pH (pH units)		6.46	6.47	6.28
Conductivity (@/cm)		12.7	15.0	13.1
Total Suspended Solids		L 1.0	L 1.0	L 1.0
Total Dissolved Solids		12.	12.	12.
Bicarbonate Alkalinity	HC03	4.21	4.91	4.91
Carbonate Alkalinity	C03	Nil	Nil	Nil
Ortho Phosphorus	P	0.007	0.011	0.010
Total Dissolved Phosphorus	P	L 0.2	L 0.2	L 0.2
Total Phosphorus	P	L 0.2	L 0.2	L 0.2
Nitrate and Nitrite Nitrogen	N	L 0.010	L 0.010	L 0.010
Ammonia Nitrogen	N	0.058	0.096	0.093
Organic Nitrogen	N	0.38	0.42	0.41
Total Kjeldahl Nitrogen	N	0.44	0.52	0.50
Total Organic Carbon	C	1.4	1.6	1.6
Total Cyanide	CN	L 0.01	L 0.01	L 0.01
<u>Total Metals</u>				
Aluminum	Al	L 0.15	L 0.15	L 0.15
Antimony	Sb	L 0.15	L 0.15	L 0.15
Arsenic	*As	0.001	0.002	0.002
Barium	Ba	0.002	0.003	0.002
Beryllium	Be	0.008	0.003	L 0.003
Bismuth	Bi	L 0.5	L 0.5	L 0.5
Boron	B	L 0.01	L 0.01	L 0.01
Cadmium	*Cd	L 0.001	L 0.001	L 0.001
Calcium	Ca	0.86	1.03	0.98
Chromium	*Cr	L 0.001	L 0.001	L 0.001
Cobalt	co	L 0.02	L 0.02	L 0.02
Copper	*cu	L 0.001	L 0.001	L 0.001
Iron	Fe	L 0.03	L 0.03	L 0.03
Lead	*Pb	L 0.001	L 0.001	L 0.001
Magnesium	Mg	0.37	0.39	0.38

Appendix A. CONT'D

Location: Outer Sun Bay, north of Narrows (cont'd)
Sample No.: EPS Site 6, 7 m depth
Date/Time: November 24, 1985, 11:45

		Lab Sample No.		
		3A	3B	3c
Manganese	Mn	0.010	0.004	0.003
Mercury	Hg	L 0.00005	L 0.00005	L 0.00005
Molybdenum	Mo	L 0.04	L 0.04	L 0.04
Nickel	*Ni	L 0.005	L 0.005	L 0.005
Phosphorus	PO4	L 0.4	L 0.4	L 0.4
Potassium	K	0.40	0.43	0.42
Silicon	SiO_2	0.14	0.18	0.17
Silver	Ag	L 0.03	L 0.03	L 0.03
Sodium	Na	0.56	0.78	0.70
Strontium	Sr	0.005	0.005	0.005
Tin	Sn	L 0.03	L 0.03	L 0.03
Titanium	Ti	L 0.006	L 0.006	L 0.006
Vanadium	V	L 0.01	L 0.01	L 0.01
Zinc	*Zn	L 0.010	L 0.010	L 0.010

All results expressed as mg/L unless noted otherwise.

* Hydride Generation or Direct AA

L = Less Than

Appendix A. CONT'D

Location: Contwoyto Lake, NE of Sun Bay
Sample No.: WQ 11, 1 m below surface
Date/Time: November 29, 1985, 11:00

		Lab Sample No.		
		6A	6B	6C
pH (pH units)		6.29	6.30	6.29
Conductivity (µS/cm)		10.5	10.5	9.66
Total Suspended Solids		L 1.0	L 1.0	L 1.0
Total Dissolved Solids		11.	9.	9.
Bicarbonate Alkalinity	HC03	4.21	4.21	4.21
Carbonate Alkalinity	co3	Nil	Nil	Nil
Ortho Phosphorus	P	0.019	0.019	0.017
Total Dissolved Phosphorus	P	L 0.02	L 0.02	L 0.02
Total Phosphorus	P	L 0.02	L 0.02	L 0.02
Nitrate and Nitrite Nitrogen	N	L 0.010	L 0.010	L 0.010
Ammonia Nitrogen	N	0.022	0.028	0.027
Organic Nitrogen	N	0.32	0.35	0.31
Total Kjeldahl Nitrogen	N	0.34	0.38	0.34
Total Organic Carbon	C	1.3	1.3	1.4
Total Cyanide	CN	L 0.01	L 0.01	L 0.01
<u>Total Metals</u>				
Aluminum	Al	L 0.15	L 0.15	L 0.15
Antimony	Sb	L 0.15	L 0.15	L 0.15
Arsenic	*As	L 0.001	L 0.001	L 0.001
Barium	Ba	0.002	0.002	0.003
Beryllium	Be	L 0.003	L 0.003	L 0.003
Bismuth	Bi	L 0.5	L 0.5	L 0.5
Boron	B	L 0.01	L 0.01	L 0.01
Cadmium	*Cd	L 0.001	L 0.001	L 0.001
Calcium	Ca	0.73	0.71	0.72
Chromium	*Cr	L 0.001	L 0.001	L 0.001
Cobalt	co	L 0.02	L 0.02	L 0.02
Copper	*cu	L 0.001	L 0.001	L 0.001
Iron	Fe	L 0.03	L 0.03	L 0.03
Lead	*Pb	L 0.001	L 0.001	L 0.001
Magnesium	Mg	0.34	0.33	0.34

Appendix A. CONT'D

Location: Contwoyto Lake, NE of Sun Bay (cont'd)
Sample No.: WQ 11, 1 m below surface
Date/Time: November 29, 1985, 11:00

		Lab Sample No.		
		6A	6B	6C
Manganese	Mn	L 0.003	L 0.003	L 0.003
Mercury	Hg	L 0.00005	L 0.00005	L 0.00005
Molybdenum	Mo	L 0.04	L 0.04	L 0.04
Nickel	*Ni	L 0.005	L 0.005	L 0.005
Phosphorus	PO4	L 0.4	L 0.4	L 0.4
Potassium	K	0.36	0.36	0.36
Silicon	SiO_2	L 0.08	L 0.08	L 0.084
Silver	Ag	L 0.03	L 0.03	L 0.03
Sodium	Na	0.38	0.33	0.36
Strontium	Sr	L 0.001	L 0.001	L 0.001
Tin	Sn	L 0.03	L 0.03	L 0.03
Titanium	Ti	L 0.006	L 0.006	L 0.006
Vanadium	V	L 0.01	L 0.01	L 0.01
Zinc	*Zn	L 0.010	L 0.010	L 0.010

All results expressed as mg/L unless noted otherwise.

* Hydride Generation or Direct AA

L = Less Than

Appendix A. CONT'D

Location: Contwoyto Lake, NE of Sun Bay
Sample No.: WQ 11, 12 m
Date/Time: November 29, 1985, 11:00

		Lab Sample No.		
		/A	7B	7C
pH (ph units)		6.34	6.28	6.27
Conductivity (µS/cm)		11.2	11.3	11.8
Total Suspended Solids		L 1.0	L 1.0	L 1.0
Total Dissolved Solids		10.	10.	10.
Bicarbonate Alkalinity	HC03	4.21	4.21	4.21
Carbonate Alkalinity	co3	Nil	Nil	Nil
Ortho Phosphorus	P	0.014	0.019	0.019
Total Dissolved Phosphorus	P	L 0.02	L 0.02	L 0.02
Total Phosphorus	P	L 0.02	L 0.02	L 0.02
Nitrate and Nitrite Nitrogen	N	L 0.010	L 0.010	L 0.010
Ammonia Nitroyen	N	0.049	0.039	0.049
Organic Ni trogen	N	0.25	0.15	0.10
Total Kjeldahl Nitrogen	N	0.30	0.19	0.15
Total Organic Carbon	C	1.3	1.4	1.3
Total Cyanide	CN	L 0.01	L 0.01	L 0.01
Total Metals				
Aluminum	Al	L 0.15	L 0.15	L 0.15
Antimony	Sb	L 0.15	L 0.15	L 0.15
Arsenic	"As	L 0.001	L 0.001	L 0.001
Barium	Ba	0.002	0.003	0.002
Beryllium	Be	L 0.003	L 0.003	L 0.003
Bismuth	Bi	L 0.5	L 0.5	L 0.5
Boron	B	L 0.01	L 0.01	L 0.01
Cadmium	*Cd	L 0.001	L 0.001	L 0.001
Calcium	Ca	0.71	0.71	0.72
Chromium	*Cr	L 0.001	L 0.001	L 0.001
Cobalt	co	L 0.02	L 0.02	L 0.02
Copper	*cu	L 0.001	L 0.001	L 0.001
Iron	Fe	L 0.03	L 0.03	L 0.03
Lead	*Pb	L 0.001	L 0.001	L 0.001
Magnesium	My	0.33	0.33	0.33

Appendix A. CONT'D

Location: Contwoyto Lake, NE of Sun Bay (cont'd)
Sample No.: WQ 11, 12 m
Date/Time: November 29, 1985, 11:00

		Lab Sample No.		
		7A	7B	7c
Manganese	Mn	L 0.003	L 0.003	L 0.003
Mercury	Hg	L 0.00005	L 0.00005	L 0.00005
Molybdenum	Mo	L 0.04	L 0.04	L 0.04
Nickel	*Ni	L 0.005	L 0.005	L 0.005
Phosphorus	PO4	L 0.4	L 0.4	L 0.4
Potassium	K	0.35	0.35	0.35
Silicon	SiO_2	0.084	0.11	0.41
Silver	Ag	L 0.03	L 0.03	L 0.03
Sodium	Na	0.36	0.38	0.41
Strontium	Sr	L 0.005	L 0.005	L 0.004
Tin	Sn	L 0.03	L 0.03	L 0.03
Titanium	Ti	L 0.006	L 0.006	L 0.006
Vanadium	V	L 0.01	L 0.01	L 0.01
Zinc	*Zn	L 0.010	L 0.010	L 0.010

All results expressed as my/L unless noted otherwise.

* Hydride Generation or Direct AA

L = Less Than

Appendix A. CONT'D .

Location: Inner Sun Bay
Sample No.: EPS Site 8, mid-depth 1.5 m
Date/Time: November 29, 1985, 17:00

		Lab Sample No.		
		8A	8B	Distilled Water Blank
pH (pH units)		6.44	6.20	
Conductivity (µS/cm)		59.2	59.1	
Total Suspended Solids		L 1.0	L 1.0	
Total Dissolved Solids		54.	55.	
Bicarbonate Alkalinity	HC03	8.42	6.31	
Carbonate Alkalinity	co3	Nil	Nil	
Total Metals				
Aluminum	Al	L 0.15	L 0.15	L 0.15
Antimony	Sb	L 0.15	L 0.15	L 0.15
Arsenic	*As	0.018	0.018	L 0.001
Barium	Ba	0.005	0.005	L 0.001
Beryllium	Be	L 0.003	L 0.003	L 0.003
Bismuth	Bi	L 0.5	L 0.5	L 0.5
Boron	B	L 0.01	L 0.010	L 0.01
Cadmium	*Cd	L 0.025	L 0.025	L 0.001
Calcium	Ca	3.56	3.52	0.003
Chromium	*Cr	L 0.03	L 0.03	L 0.001
Cobalt	co	L 0.02	L 0.02	L 0.02
Copper	*cu	L 0.001	L 0.001	L 0.001
Iron	Fe	0.18	0.18	L 0.03
Lead	*Pb	L 0.001	L 0.001	L 0.001
Magnesium	Mg	0.61	0.61	L 0.002
Manganese	Mn	0.008	0.008	L 0.003
Mercury	Hg			L 0.00005
Molybdenum	Mo	L 0.04	L 0.04	L 0.04
Nickel	*Ni	L 0.005	L 0.005	L 0.005
Phosphorus	P04	L 0.4	L 0.4	L 0.4
Potassium	K	0.63	0.71	L 0.01
Silicon	Si02	0.69	0.67	L 0.08
Silver	Ag	L 0.03	L 0.03	L 0.03
Sodium	Na	4.18	4.14	0.10
Strontium	Sr	0.012	0.011	L 0.001
Tin	Sn	L 0.03	L 0.03	L 0.03
Titanium	Ti	L 0.006	L 0.006	L 0.006
Vanadium	V	L 0.01	L 0.01	L 0.01
Zinc	*Zn	L 0.010	L 0.010	L 0.010

All results expressed as mg/L unless noted otherwise.
* Hydride Generation or Direct AA

Appendix A. CONT'D

Location: Mouth of Outer Sun Bay (by island)
Sample No.: WQ 10, #4 = 1 m; #5 = 7 m
Date/Time: November 24, 1985, 15:00

		Lab Sample No.			
		4A	4B	5A	5B
pH (pH units)			6.33	6.33	
Conductivity (µS/cm)			11.9	11.8	
Total Suspended Solids			L 1.0	L 1.0	
Total Dissolved Solids			11.	11.	
Bicarbonate Alkalinity	HC03		4.91	4.91	
Carbonate Alkalinity	co3		Nil	Nil	
Total Metals					
Aluminum	Al	L 0.15	L 0.15	L 0.15	L 0.15
Antimony	Sb	L 0.15	L 0.15	L 0.15	L 0.15
Arsenic	*As	L 0.001	L 0.001	L 0.001	L 0.001
Barium	Ba	0.003	0.003	0.002	0.002
Beryllium	Be	L 0.003	L 0.003	L 0.003	L 0.003
Bismuth	Bi	L 0.5	L 0.5	L 0.5	L 0.5
Boron	B	L 0.01	L 0.01	L 0.01	L 0.01
Cadmium	*Cd	L 0.025	L 0.025	L 0.025	L 0.025
Calcium	Ca	0.82	0.82	0.81	0.82
Chromium	*Cr	L 0.03	L 0.03	L 0.03	L 0.03
Cobalt	co	L 0.02	L 0.02	L 0.02	L 0.02
Copper	*cu	L 0.001	L 0.001	L 0.001	L 0.001
Iron	Fe	L 0.03	L 0.03	L 0.040	L 0.03
Lead	*Pb	L 0.001	L 0.001	L 0.001	L 0.001
Magnesium	Mg	0.39	0.38	0.34	0.38
Manganese	Mn	L 0.003	L 0.003	L 0.003	L 0.003
Mercury	Hg				
Molybdenum	Mo	L 0.04	L 0.04	L 0.04	L 0.04
Nickel	*Ni	L 0.005	L 0.005	L 0.005	L 0.005
Phosphorus	PO_4	L 0.4	L 0.4	L 0.4	L 0.4
Potassium	K	0.40	0.37	0.37	0.40
Silicon	SiO_2	0.10	0.11	0.14	0.096
Silver	Ag	L 0.03	L 0.03	L 0.03	L 0.03
Sodium	Na	0.40	0.41	0.53	U.46
Strontium	Sr	L 0.001	L 0.001	L 0.001	L 0.001
Tin	Sn	L 0.03	L 0.03	L 0.03	L 0.03
Titanium	Ti	L 0.006	L 0.006	L 0.006	L 0.006
Vanadium	V	L 0.01	L 0.01	L 0.01	L 0.01
Zinc	*Zn	L 0.010	L 0.010	L 0.012	L 0.010

All results expressed as mg/L unless noted otherwise.
* Hydride Generation or Direct AA
L = Loss Than

APPENDIX B

APPENDIX B. Fish tissue analysis for Contwoyto Lake, N.W.T. in fall, 1985. Metal concentrations are expressed as ppm on a wet weight basis.

									Metals (ppm wet weight)					
Location	Date	Species	Sample #	Tissue THPC	Length (mm)	Weight (gm)	Sex Code	Moisture (%)	As	Cd	Pb	Hg	Ni	Zn
ESC[1]	30/9/85	LT	2086	L	541	1770	11	76.2	0.01	0.43	0.01	0.15	0.15	26.42
ESC	1/10/85	LT	2100	L	534	1575	1	-[2]	0.01	0.63	0.01	0.08	0.12	34.70
ESC	2/10/85	LT	2107	L	681	3040	9	76.0	0.01	0.48	0.01	0.48	0.23	40.56
ESC	2/10/85	LT	2108	L	424	530	9		0.01	0.53	0.01	0.14	0.14	29.60
ESC	2/10/85	LT	2116	L	541	1680	9	80.7	0.00	0.38	0.00	0.15	0.10	26.25
ESC	30/9/85	LT	2086	M	541	1770	11	79.5	0.01	0.01	0.01	0.17	0.03	4.86
ESC	1/10/85	LT	2100	M	534	1575	1	76.5	0.01	0.01	0.01	0.10	0.03	5.33
ESC	2/10/85	LT	2107	M	681	3040	9	77.3	0.01	0.01	0.01	0.26	0.19	5.52
ESC	2/10/85	LT	2108	M	424	530	9	79.5	0.01	0.01	0.01	0.11	0.09	4.55
ESC	2/10/85	LT	2116	M	541	1680	9	79.1	0.01	0.00	0.01	0.14	0.03	4.20
SB	28/19/85	LT	2003	L	553	1990	0	75.3	0.01	0.21	0.01	0.22	0.22	24.65
SB	29/9/85	LT	2034	L	561	1750	9		0.01	0.45	0.01	0.16	0.11	38.90
SB	29/9/85	LT	2036	L	552	1825	19		0.01	0.61	0.01	0.21	0.10	36.10
SB	29/9/85	LT	2060	L	541	2130	9	79.2	0.01	0.44	0.01	0.22	0.23	23.92
SB	29/9/85	LT	2063	L	541	1700	19	80.0	0.10	0.49	0.01	0.29	0.15	44.60
SB	24/11/85	LT	2122	L	582	1905	19	78.5	0.01	0.29	0.01	0.26	0.13	36.55
SB	24/11/85	LT	2122	L	563	2055	9	78.3	0.01	0.42	0.01	0.22	0.18	29.51
SB	25/11/85	LT	2123	L	578	1955	18	71.6	0.01	0.32	0.01	0.15	0.10	28.97
SB	29/11/85	LT	2152	L	530	1480	0		0.01	0.58	0.01	0.26	0.41	40.20
SB	29/11/85	LT	2154	L	540	1460	19	79.9	0.12	0.45	0.01	0.23	0.26	31.36
SB	29/19/85	LT	2003	M	553	1990	0	75.7	0.02	0.01	0.01	0.17	0.03	5.30
SB	29/9/85	LT	2034	M	561	1750	9	77.3	0.01	0.01	0.01	0.21	0.03	5.99
SB	29/9/85	LT	2036	M	552	1825	19	75.0	0.03	0.01	0.01	0.17	0.03	4.97
SB	29/9/85	LT	2060	M	541	2130	9	76.6	0.01	0.01	0.01	0.18	0.03	4.59
SB	29/9/85	LT	2063	M	541	1700	19	76.4	0.09	0.01	0.01	0.16	0.07	6.23
SB	24/11/85	LT	2121	M	582	1905	19	77.1	0.01	0.01	0.01	0.27	0.03	4.31
SB	24/11/85	LT	2122	M	563	2055	9	79.6	0.01	0.01	0.01	0.21	0.03	5.14
SB	24/11/85	LT	2123	M	578	1955	18	76.6	0.01	0.01	0.01	0.17	0.03	5.57
SB	29/11/85	LT	2152	M	530	1480	0	78.2	0.03	0.01	0.11	0.21	0.03	3.66
SB	29/11/85	LT	2154	M	540	1460:	19	81.4	0.03	0.00	0.00	0.24	0.02	4.18
SB	24/11/85	RW	2132	L	436	830	0		1.40	0.52	0.01	0.22	1.01	66.50
SB	25/11/85	RW	2126	L	452	925	0		0.01	0.31	0.01	0.19	0.62	85.00
SB	29/11/85	RW	2153	L	440	870	0		0.31	0.19	0.01	0.14	0.50	23.70
SB	29/11/85	RW	2149	L	430	920	0		1.74	0.10	0.01	0.16	0.03	16.00
SB	25/11/85	RW	2127	L	399	690	0		0.01	0.17	0.01	0.15	0.83	91.70
SB	24/11/85	RW	2132	M	436	830	0	75.1	0.05	0.01	0.01	0.06	0.03	6.27
SB	25/11/85	RW	2126	M	452	925	0	76.6	0.01	0.01	0.01	0.10	0.03	6.44
SB	29/11/85	RW	2153	M	440	870	0	77.1	0.01	0.01	0.01	0.09	0.09	5.59
SB	29/11/85	RW	2149	M	430	920	0	78.0	0.03	0.01	0.01	0.14	0.03	5.10
SB	25/11/85	RW	2127	M	399	690	0	76.6	0.03	0.01	0.01	0.13	0.03	6.11

[1] Abbreviations: ESC = East Side Contwoyto Lake; SB = Inner Sun Bay; Species; LT = Lake Trout; RW = Round Whitefish; L = liver; M = muscle; Sex Code (0 = not determined; 1-10 = male; 11 - 20 = female).

[2] Liver tissue samples with no moisture values were analyzed on "wet weight as received basis" due to insufficient sample Size.

APPENDIX C

APPENDIX C. Government agencies contacted reyarding metal contamination data.

AGENCY	LOCATION	CONTACT	DATA
FEDERAL			
1. Environmental Protection Service, Dept. of Environment	Whitehorse, Yukon	George MacKenzie-Grieve	Lake Laberge, Little Gold Creek, Cyprus Anvil
	Yellowknife, N.W.T.	Dave Sutherland	Rayrock, Port Radium, Con Mine, Back Bay, Great Slave Lake, Discovery, Canada Tungsten, Giant Yellowknife
	Vancouver, B.C.	Benoit Godin	Francois Lake, Babine Lake
2. Dept. of Indian Affairs & Northern Development	Yellowknife N.W.T.	Ian Sharp (Doug Stendahl)	No data
3. Dept. of Fisheries & Oceans	Whitehorse, Yukon	Pete Heatherton	Fish metal (Hg) data, Lake Laberge
a. Freshwater Institute	Winnipeg, Manitoba	Bob Hecky, Dave Schindler, Jack Klaverkamp	South Indian Lake; metals, radio nucleide - Experimental Lakes Area
4. Inland Waters Directorate Dept. of the Environment			
a. Canada Centre for Inland Waters	Burlington, Ontario	Gerome Nriagu	Environmental impact of smelters Contaminant effects on fish
b. National Water Research Institute	Vancouver, B.C.	Chris Pharo	Hg in fish - Thompson River
	Burlington, Ontario	Dr. A. Mudroch	Lakes Three Mile, Powder Mill, Thomas Third, Fletcher, Perry and Muddy Pond, N.S. were sampled for sediment, water and fish contamination in conjunction with Nova Scotia Department of the Environment

APPENDIX C. Continued

AGENCY	LOCATION	CONTACT	DATA
PROVINCIAL			
1. British Columbia			
a. Ministry of Environment Water Management Branch	Victoria	Colin McKean	Computerized data set for over 100 lakes with sediment and water quality; 40 lakes also have fish data
Waste Management Branch	Victoria	Malcolm Clark	Buttle Lake and Campbell River drainage system reports.
	Smithers	Brian Wilkes	Aldrich Lake
	Prince George	Rich Girard	Lakes of the Pinchi fault area of B.C., with water, fish and sediments data and some shellfish data
	Nelson	Rick Crozier	Kootenay Lake - water quality and fish data
2. Alberta			
a. Pollution Control Branch	Edmonton	Akio Masuda	No data
b. Environmental Research Centre	Vegfeville	Jim Moore	Hg partitioning N. Saskatchewan River
3. Saskatchewan			
a. Saskatchewan Environment	Regina	Bob Ruggles	Data not yet available for public distribution

APPENDIX C. Continued

AGENCY	LOCATION	CONTACT	DATA
PROVINCIAL (cont'd)			
4. Manitoba			
a. Environment & Workplace Safety & Health,. Environmental Management	Winnipeg	Dennis J. Brown	Four reports on 6 lakes with chemistry, fish and sediments for heavy metals. Interesting biomonitoring programing using fish and sediments in river systems
5. Ontario			
1. Ministry of Environment Water Resource Centre	Dorset	Peter Dillon	No data
6. Quebec			
a. Ministry of Environment	Sainte-Foy	Denis Laliberte	Extensive river data with water, sediment, plant, invertebrates and fish (computerized), not much lake data