AQUATIC EFFECTS TECHNOLOGY EVALUATION (AETE) PROGRAM

Summary and Cost-Effectiveness Evaluation of Aquatic Effects Monitoring Technologies Applied in the 1997 AETE Field Evaluation Program

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SUMMARY AND COST-EFFECTIVENESS EVALUATION OF AQUATIC EFFECTS MONITORING TECHNOLOGIES APPLIED IN THE 1997 AETE FIELD EVALUATION PROGRAM

Report prepared for:

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AQUATIC EFFECTS TECHNOLOGY EVALUATION PROGRAM

Notice to Readers

Summary and Cost-Effectiveness Evaluation of Aquatic Effects Monitoring Technologies Applied in the 1997 AETE Field Evaluation Program

The Aquatic Effects Technology Evaluation (AETE) program was established to review appropriate technologies for assessing the impacts of mine effluents on the aquatic environment. AETE is a cooperative program between the Canadian mining industry, several federal government departments and a number of provincial governments; it is coordinated by the Canada Centre for Mineral and Energy Technology (CANMET). The program was designed to be of direct benefit to the industry, and to government. Through technical and field evaluations, it identified cost-effective technologies to meet environmental monitoring requirements. The program included three main areas: acute and sublethal toxicity testing, biological monitoring in receiving waters, and water and sediment monitoring.

The technical evaluations are conducted to document certain tools selected by AETE members, and to provide the rationale for doing a field evaluation of the tools or provide specific guidance on field application of a method. In some cases, the technical evaluations included a go/no go recommendation that AETE takes into consideration before a field evaluation of a given method is conducted.

The technical evaluations are published although they do not necessarily reflect the views of the participants in the AETE Program. The technical evaluations should be considered as working documents rather than comprehensive literature reviews. The purpose of the technical evaluations was to document specific monitoring tools. AETE committee members would like to stress that no one single tool can provide all the information required for a full understanding of environmental effects in the aquatic environment.

For more information on the monitoring techniques, the results from their field application and the final recommendations from the program, please consult the AETE Synthesis Report to be published in the spring of 1999.

Any comments concerning the content of this report should be directed to:

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PROGRAMME D'ÉVALUATION DES TECHNIQUES DE MESURE D'IMPACTS EN MILIEU AQUATIQUE

Avis aux lecteurs

Sommaire et évaluation du rapport coût-efficacité des méthodes de surveillance des effets des effluents miniers sur les écosystèmes aquatiques évaluées sur le terrain en 1997 dans le cadre du programme ÉTIMA

Le Programme d'évaluation des techniques de mesure d'impacts en milieu aquatique (ÉTIMA) visait à évaluer les différentes méthodes de surveillance des effets des effluents miniers sur les écosystèmes aquatiques. Il est le fruit d'une collaboration entre l'industrie minière du Canada, plusieurs ministères fédéraux et un certain nombre de ministères provinciaux. Sa coordination relève du Centre canadien de la technologie des minéraux et de l'énergie (CANMET). Le programme était conçu pour bénéficier directement aux entreprises minières ainsi qu'aux gouvernements. Par des évaluations techniques et des études de terrain, il a permis d'évaluer et de déterminer, dans une perspective coût-efficacité, les techniques qui permettent de respecter les exigences en matière de surveillance de l'environnement. Le programme comportait les trois grands volets suivants : évaluation de la toxicité aiguë et sublétale, surveillance des effets biologiques des effluents miniers en eaux réceptrices, et surveillance de la qualité de l'eau et des sédiments.

Les évaluations techniques ont été menées dans le but de documenter certains outils de surveillance sélectionnés par les membres de l'ÉTIMA et de fournir une justification pour l'évaluation sur le terrain de ces outils ou de fournir des lignes directrices quant à leur application sur le terrain. Dans certains cas, les évaluations techniques pourraient inclure des recommandations relatives à la pertinence d'effectuer une évaluation de terrain que les membres de l'ÉTIMA prennent en considération.

Les évaluations techniques sont publiées bien qu'elles ne reflètent pas nécessairement toujours l'opinion des membres de l'ÉTIMA. Les évaluations techniques devraient être considérées comme des documents de travail plutôt que des revues de littérature complètes. Les évaluations techniques visent à documenter des outils particuliers de surveillance. Toutefois, les membres de l'ÉTIMA tiennent à souligner que tout outil devrait être utilisé conjointement avec d'autres pour permettre d'obtenir l'information requise pour la compréhension intégrale des impacts environnementaux en milieu aquatique.

Pour des renseignements sur l'ensemble des outils de surveillance, les résultats de leur application sur le terrain et les recommandations finales du programme, veuillez consulter le Rapport de synthèse ÉTIMA qui sera publié au printemps 1999.

Les personnes intéressées à faire des commentaires concernant le contenu de ce rapport sont invitées à communiquer avec M^{me} Geneviève Béchard à l'adresse suivante :

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1.0 INTRODUCTION

The Assessment of the Aquatic Effects of Mining in Canada (AQUAMIN), initiated in 1993, evaluated the effectiveness of Canada's *Metal Mining Liquid Effluent Regulations* (MMLER). One of the key recommendations of the 1996 AQUAMIN Final Report is that a revised MMLER include a requirement that metal mines conduct Environmental Effects Monitoring (EEM), to evaluate the effects of mining activity on the aquatic environment, including fish, fish habitat and the use of fisheries resources.

In parallel, the Canada Centre for Mineral and Energy Technology (CANMET) is coordinating a cooperative government-industry program, the Aquatic Effects Technology Evaluation (AETE) program, to review and evaluate technologies for the assessment of mining-related impacts in the aquatic environment. The intention of the AETE program is to evaluate and identify cost-effective technologies to meet environmental monitoring requirements at mines in Canada. The program is focused on evaluation of environmental monitoring tools that may be used for a national mining EEM program, baseline assessments or general impact studies.

The three principal components of the AETE program are lethal and sublethal toxicity testing of water/effluents and sediments, biological monitoring in receiving waters, and water and sediment chemistry assessments. The program includes both literature-based technical evaluations and comparative field programs at candidate sites.

An AETE Pilot Field Study was carried out in the Val d'Or region of Québec in 1995 to evaluate several environmental monitoring methods and to reduce the list of monitoring technologies for further evaluation at a cross-section of mine sites across Canada (BEAK, 1996). In 1996, a field evaluation program was initiated which involved preliminary sampling at seven candidate mine sites with the objective of identifying a short-list of mines that had suitable conditions for further detailed monitoring and testing of hypotheses related to the AETE program. Preliminary study designs were developed for four sites that were deemed to be most suitable for hypotheses testing in 1997 (EVS *et al.*, 1997). The sites selected were Heath Steele, New Brunswick; Lupin, Northwest Territories; Dome Mine, Ontario; and Westmin Resources (now Boliden-Westmin), Myra Falls, British Columbia. Lupin was subsequently dropped based on a 1997 reconnaissance survey and replaced with the Mattabi Mines Ltd. site in Ignace, Ontario (BEAK and GOLDER, 1998a). The 1997 program consisted of a review of necessary background information, finalization of study designs, implementation of the field studies and hypotheses testing and reporting of results.

1.1 Study Objectives

The overall goal of the AETE Program is to identify cost-effective methods and technologies that are suitable for assessing aquatic environmental effects caused by mining activity. An effect is defined as “a measurable difference in an environmental variable (chemical, physical or biological) between a point downstream (or exposed to mining) in the receiving environment and an adequate reference point (either spatial or temporal)”. Based on this definition, the AETE Technical Committee developed a series of hypotheses to be tested under field conditions at a number of mine sites in Canada. The Committee agreed that specific hypotheses should be articulated in order to clarify the purpose of the program elements. For the formulation of the hypotheses, the definition of an effect was refined by the AETE Technical Committee to distinguish between effects or responses as measured in biological variables as opposed to effects reflected in physical or chemical changes.

The questions used in developing the hypotheses to be tested in the 1997 field evaluation program were:

1. **Are contaminants getting into the system (and to what degree, and in which compartments)?** This question relates to the presence of elevated concentrations of metals in environmental media (e.g., water, sediments), and requires an understanding of metal dispersal mechanisms, chemical reactions in sediment and water, and aquatic habitat features which influence exposure of biological communities.

2. **Are contaminants bioavailable?** This question relates to the presence of metals in biota or to indicators of bioaccumulation, such as the induction of metallothionein in fish. Only if contaminants are bioavailable can a biological effect from chemical contaminants occur.

3. **Is there a measurable response?** Biological responses may occur only if contaminants are entering the environment and occur in bioavailable forms. These responses may occur at various levels of biological organization, including sub-organism levels (e.g., histopathological effects), at the organism level (e.g., as

measured in toxicity testing), or at population and community levels (e.g., as measured in resident benthic invertebrate and fish communities).

4. **Are contaminants causing the responses?** This question is difficult to answer in field studies directly, as cause-effect mechanisms are difficult to assess under variable conditions prevailing in nature. However, correlations between measures of exposure, chemical bioavailability and response may be used to develop evidence useful in evaluating this question.

The AETE Technical Committee developed a study framework, using the above questions and the three components (water and sediment monitoring, biological monitoring in receiving waters and toxicity testing). The following eight areas of work were identified to finalize the work plan, develop the hypotheses, prioritize issues and identify field work requirements:

1. Chemical presence;
2. The overlap between communities and chemistry testing to determine whether biological responses are related to a chemical presence (bioavailability of contaminants);
3. Biological response in the laboratory;
4. Biological response in the field;
5. Chemical characteristics of the water and sediments used to predict biological responses in the field (contaminants causing a response);
6. The overlap between biological community responses and bioassay responses to evaluate whether community changes in the field are predicted by bioassay responses;
7. The overlap between chemistry and bioassay responses to evaluate whether chemicals are responsible for bioassay responses; and
8. The overlap between the chemical, the exposure and the effects in the laboratory and the effects in the field.

The core objective, however, was **to test the 13 hypotheses, developed by the AETE Committee, at as many of the four selected mine sites as possible (Table 1.1).** The hypotheses are more specific questions about the ability or relative ability of different monitoring tools to answer the four general questions (above) about mine effects.

TABLE 1.1: HYPOTHESES TESTED IN 1997. AETE FIELD PROGRAM

Sediment Monitoring

H1. Sediment Toxicity:

H: *The strength of the relationship between sediment toxicity responses and any exposure indicator is not influenced by the use of different sediment toxicity tests or combinations of toxicity tests.*

Biological Monitoring - Fish

H2. Metals in Fish Tissues (bioavailability of metals):

H: *There is no difference in metal concentrations observed in fish liver, kidney, gills, muscle or viscera.*

H3. Metallothionein in Fish Tissues:

H: *There is no difference in metallothionein concentration observed in liver, kidney, gills, viscera*

H4. Metal vs. Metallothionein in Fish Tissues:

H: *The choice of metallothionein concentration vs. metal concentration in fish tissues does not influence the ability to detect environmental exposure of fish to metals.*

H5. Fish - CPUE:

H: *There is no environmental effect in observed CPUE (catch per unit effort) of fish.*

H6. Fish (or Benthic) - Community:

H: *There is no environmental effect in observed fish community structure.*

H7: Fish - Growth:

H: *There is no environmental effect in observed fish growth.*

H8. Fish - Organ/Fish Size:

H: *There is no environmental effect in observed organ size (or fish size, etc.).*

Integration of Tools

H9. Relationship Between Water Quality and Biological Components:

H: *The strength of the relationship between biological variables and metal chemistry in water is not influenced by the choice of total vs. dissolved analysis of metals concentration.*

H10. Relationship Between Sediment Chemistry and Biological Responses:

H: *The strength of the relationship between biological variables and sediment characteristics is not influenced by the analysis of total metals in sediments vs. either metals associated with iron and manganese oxyhydroxides or with acid volatile sulphides.*

H11. Relationship Between Sediment Toxicity and Benthic Invertebrates:

H: *The strength of the relationship between sediment toxicity responses and in situ benthic macroinvertebrate community characteristics is not influenced by the use of different sediment toxicity tests, or combinations of toxicity tests.*

H12. Metals or Metallothionein vs. Chemistry (receiving water and sediment):

H: *The strength of the relationship between the concentration of metals in the environment (water and sediment chemistry) and metal concentration in fish tissues is not different from the relationship between metal concentration in the environment and metallothionein concentration in fish tissues.*

H13. Chronic Toxicity - Linkage with Fish and Benthos Monitoring Results:

H: *The suite of sublethal toxicity tests cannot predict environmental effects to resident fish performance indicators or benthic macroinvertebrate community structure.*

These 13 hypotheses can be categorized into:

- ***Sediment Monitoring***: evaluation of sediment toxicity testing tools (test types) as to their relative ability to detect linkages between mine exposure and sediment toxicity (H1);

- ***Biological Monitoring (in Fish)***: evaluation of tissue biomonitoring tools (measurement types) as to their ability to detect linkages between mine exposure and tissue contamination (H2 to H4); and evaluation of population/community biomonitoring tools (measurement types) as to their ability to detect linkages between mine exposure and ecological response (H5 to H8); and

- ***Integration of Tools***: evaluation of various monitoring tools as to their relative ability to detect relationships between specific measures of mine exposure and specific biological response measures, or between sediment toxicity and benthic community response measures (H9 to H12); and evaluation of effluent toxicity testing tools (test types) as to their ability to detect relationships between effluent toxicity and population/community response measures (H13).

Detailed methods and results of hypotheses testing for each of the four selected mine sites are presented in separate site reports (BEAK, 1998a,b; BEAK and Golder, 1998b,c). This document summarizes the results from those reports and compares the overall cost-effectiveness of each of the monitoring tools, based on the results from all of the sites.

For the most part, each hypothesis was generally tested at three of the four mine sites, because conditions were often sub-optimal from a study design perspective to test all hypotheses at all sites. The reader is cautioned not to assume that the conclusions drawn from the results of tool evaluations at three mines are necessarily broadly valid at all mine sites across Canada. Indeed, the results of these studies clearly show that monitoring tools vary considerably in their performance among the 1997 field studies and tool performance is often influenced by confounding factors (e.g., variability in habitats, other sources of contamination). This variation in monitoring tool performance may also be attributed to other factors, such as differences in metal concentrations and metal mixtures in the environment; differences in metal bioavailability from mine to mine; and differences in responses due to the use of different biological tools (species, communities) at each mine. However, at all four mine sites some of the tools (e.g., benthic invertebrates) showed that there were mine effects. Therefore, the results of this study do provide a good indication of the nature of mine-related effects on aquatic communities and on monitoring tool performance in a controlled and comparative sense under a cross-section of mining and environmental conditions.

1.2 Site Descriptions

1.2.1 Heath Steele

Heath Steele Division of Noranda Mining and Exploration Inc. (Heath Steele) operates a base metal mining and milling operation in north-central New Brunswick, approximately 50 km northwest of the city of Miramichi. Mine/mill operations are situated within the headwaters of the Tomogonops River, a tributary system of the Northwest Miramichi River (Figure 1.1).

The Heath Steele site has a relatively long history, with mine and mill facilities first developed in 1955 to 1957. Heath Steele ores are base metal sulphides, with zinc, lead, copper and silver-rich concentrates produced.

The South Branch Tomogonops River receives discharge from the tailings area, but this stream has, in recent years, become periodically acidic due to thiosalt oxidation and is high in dissolved solids (BEAK, 1997). This has produced a relatively strong pH gradient in the South Branch Tomogonops River, especially in summer. The metal concentration gradient in the South Branch is relatively weak (small changes with distance), and fish are scarce in the reach nearest the tailings pond.

The Little South Branch Tomogonops River receives seepage and runoff from the general mine site that is not strongly acidic and the water is much softer than treated effluent. These characteristics occur at Heath Steele monitoring Station HS-3, downstream of which no significant additional inputs occur from Heath Steele. This water is relatively rich in metals, especially zinc, copper, cadmium and lead.

Downstream gradients in water quality and biological conditions have been well documented in the Little South Branch Tomogonops and main Tomogonops River (BEAK, 1997).

Accordingly, the 1997 AETE field program focused on river reaches in the Little South Branch Tomogonops River and waters downstream before the confluence with the South Branch Tomogonops, where water hardness level abruptly increases. Concentrations of "effluent", designated as Little South Branch Tomogonops water at HS-3, ranged from 11% in the far-field study area to 60% in the near-field study area.

A railway bridge across the Tomogonops River at times presents a barrier to upstream migration of adult salmon. Therefore, the fish community is different below the bridge compared to above. Fish present above and below the bridge include juvenile salmon, brook trout, white sucker and minnows, although the abundance of salmon upstream of the bridge appears to be reduced by the barrier. An apparently fishless zone exists immediately below the mine at HS-3, apparently due to water quality impairment.

Aquatic habitat throughout the study area consists of riffles and runs, with a predominantly rock-cobble-gravel streambed. Soft sediments are rare to absent throughout most of the Tomogonops River watershed. The predominant erosional condition of the river prevents effective testing of sediment monitoring tools at Heath Steele. The watershed is undeveloped and forested except for the mine site itself. The streamflow was low ($\leq$0.31 m^3/s) at all locations sampled in August 1997, with typical stream widths of up to about 8 m. Stream size is progressively smaller towards upper reaches of the watershed. All reference areas selected for study, including the neighbouring Little River and unaffected reaches of the Tomogonops River, are similar to those represented by the area of downstream habitat sampled, except for the partial barrier noted above.

1.2.2 Dome Mine

The Dome Mine is one of the largest underground/open pit gold mines in Canada and is located in South Porcupine, near Timmins, Ontario. The operations which started in 1910 represent one of the oldest and largest mines in Canada.

The South Porcupine River is the receiving environment for mine effluent discharged from Dome's #6 Dam (Figure 1.2). The river is a low-gradient, mud-bottom stream with dense macrophytes throughout its length. Some sections are almost two metres deep because of a number of beaver dams along the creek. The effluent is fully mixed with receiving water within 500 m, and the North Porcupine River approximately 3 km downstream adds substantial dilution water. About 2 km downstream from the confluence of the two branches, the Porcupine River flows into Porcupine Lake. There are several abandoned mines and mine waste areas along the South Porcupine River, upstream of Dome Mine, that influence water quality (Figure 1.2).

Discharge from the #6 Dam is largely seasonal, and at times is treated by an INCO-SO_2/Air cyanide destruction process before release to receiving waters. The operation utilizes gravity settling to produce a clear pond where effluent is recycled back to the mill for reuse. Excess effluent is treated using best available technology economically achievable (BATEA) prior to discharge. The INCO treatment system is only used when natural cyanide degradation is inadequate. During the discharge period, the estimated effluent concentration in the receiving waters, based on flow estimates provided by Dome, is an average of 37% effluent in the area upstream of the confluence with the North Porcupine River, and 16% from the confluence downstream to Porcupine Lake. In October 1997, during the field survey, the mine was not discharging effluent (effluent discharge ceased on 12 August 1997).

The predominant contaminants of concern at the Dome Mine are copper, nickel, cobalt, selenium and silver.

The pattern of effluent dilution at the Dome Mine site determined the study design, with a reference reach upstream of the discharge, and two exposure reaches downstream: a near-field reach above the confluence of the North Porcupine, and a far-field reach below the confluence (for benthos only, no fish could be captured in this reach). Since lake conditions have distinct influences on biological communities, Porcupine Lake fish communities (exposure area) cannot be compared to those in the stream, so a separate lake reference area for fish was established in McDonald’s Lake, the upstream source of the South Porcupine River.

Owing to the extensive historic mining disturbance in the area, reference areas were not free of mine-related contaminants, but were sited as far as possible from historic tailings within the constraints imposed by existing hydrology and natural setting.

1.2.3 Mattabi Mine

Mattabi Mines Ltd. (Mattabi) operated a copper-lead-zinc open pit and underground mining operation and concentrator complex between 1972 and 1991 approximately 80 km northeast of Ignace, northwestern Ontario, between Sturgeon Lake and Bell Lake (Figure 1.3). The mine was developed between 1970 and 1972 and produced ore between 1972 and 1988. Milling operations continued until 1991. Operations also included mining at the F-Group open pit (development 1979-1980, production 1980-1984) 9 km west of the Mattabi Mine site and the Lyon Lake Mine (Lyon Lake Division; development 1974-1980, production 1980-1991), about 13 km east of the site. Ores from Lyon Lake and F-Group were milled at the Mattabi concentrator.

Rehabilitation activities continue at Mattabi, and effluent continues to be treated and discharged to the environment from a treatment plant. Effluent is discharged intermittently to Bell Creek, which flows into Sturgeon Lake. Effluent was being discharged at the time of the Mattabi field survey, producing an estimated concentration of about 2% in Bell Creek.

Previous studies at the site indicated that there are impacts from the tailings and polishing pond areas on Bell Creek water, sediments and benthos. These effects extend 6 km downstream of the polishing pond discharge, with some recovery evident upstream of the mouth of Bell Creek into Bell Creek Bay (Sturgeon Lake). The principal contaminants are zinc, copper, lead and cadmium. In addition, the impact of Mattabi on Mine Creek Bay (Sturgeon Lake) sediment metal concentrations (zinc, copper, lead and cadmium) is limited to an approximate 400 m radius from the outlet of Mine Creek. The benthic community within this area showed some impairment from contamination.

Based on previous studies, concentrations of zinc and other metals are known to be elevated in Bell Creek water downstream of Mattabi. Metal concentrations in sediments of Bell Creek, No Name Lake and Mine Creek Bay in 1991-92 were consistent with those reported in 1989, with the depth of contamination extending about 10 cm below the sediment surface. The most contaminated sediments were found in No Name Lake.

Previous studies have also shown that benthic densities and diversities are reduced in Bell Creek downstream of Mattabi relative to the upstream reference sites. The greatest degree of impairment has been observed in Mine Creek Bay and No Name Lake.

Bell Creek is predominantly a slow-flowing, depositional stream, although riffles are locally present downstream of the mine. The relatively wide, slow-flowing sections of stream are known to contain metal-enriched sediments. No Name Lake is a small (~20 ha), shallow (~1-2 m deep) lake in proximity to Mattabi. Mine Creek Bay is a shallow (~1-2 m deep) embayment on Sturgeon Lake. Both No Name Lake and Mine Creek Bay contain soft sediments. Reference areas selected for this study include the English River (reference for Bell Creek), Peterson Cove on Sturgeon Lake (reference for Mine Creek Bay) and Tag Lake (reference for No Name Lake).

1.2.4 Myra Falls Operations

The Myra Falls Operations of Westmin Resources (now Boliden-Westmin Myra Falls Operations) produce base metal concentrates (zinc, copper, lead), as well as silver and gold, and is located in central Vancouver Island, within Strathcona Provincial Park. The operations started in 1966 and are situated in the Myra Creek valley. Myra Creek receives metal-enriched discharges from the operations and drains into Buttle Lake, a deep, oligotrophic artificial reservoir (Figure 1.4). Buttle Lake discharges northward and eastward through a series of reservoirs into the Campbell River.

From the mid-1960s to mid-1980s, the mine discharged tailings into Buttle Lake. In response to concerns over potential impacts on aquatic resources, the mine abandoned deep lake disposal in the mid-1980s in favour of land-based disposal in an engineered tailings impoundment which lies within the Myra Creek watershed.

Metal-contaminated water from Myra Falls Operations is collected, treated with lime and polished effluent is discharged to Myra Creek. Effluent concentration in the area sampled in the creek was estimated at approximately 15%. Metal-enriched water from other sources also reaches the creek, and the data from this study indicated that zinc loadings to the creek were dominated by these other sources. The primary contaminants of concern are zinc, copper, arsenic, cadmium, lead and silver.

Myra Creek flows eastward from the mine and into the southern portion of Buttle Lake, approximately 2 km downstream. Myra Falls, a waterfall near the mouth of the creek, presents a physical barrier to the movement of fish upstream from the lake. Buttle Lake is 35 km in length with a mean depth of 45 m. Aquatic habitats in the study area include fast-flowing and erosional conditions in Myra Creek, with substrates consisting of rock and cobble. Myra Lake sediments are fine-grained, and include tailings in the southern portion of the lake. Water quality in Buttle Lake is not strongly affected by metals and concentrations are generally equal to or lower than Canadian water quality guidelines.

The area selected for the affected reach of Myra Creek was similar to habitat conditions upstream of the mine. The reference area selected for Buttle Lake was Brewster Lake, a relatively deep waterbody in the area that has not been affected by mining activity.

2.0 STUDY DESIGNS AND METHODS

2.1 Study Designs

Study designs were finalized for each of the four mine sites based on the preliminary data and study designs developed during the 1996 screening of seven mine sites (EVS *et al.,* 1997). The final study designs received AETE's approval, and were implemented from August to October 1997.

The hypotheses that could be appropriately tested at each of the mine sites are provided in Table 2.1. The tools that were evaluated in the 1997 Field Studies are provided in Table 2.2. The final study designs to address these hypotheses were based either on gradient or control-impact (CI) designs with one or two exposure areas. The designs were selected based on the natural habitat settings. In general, sampling was carried out in relation to a point source discharge in order to permit testing of hypotheses about the environmental effect of the discharge. Sampling was carried out both above and below the source (Reference versus Exposure).

Gradient designs were applied to all study components at Heath Steele and to the sediment monitoring components (i.e., sediment chemistry, sediment toxicity, benthic invertebrates) at Mattabi Mine. The gradient designs had numerous "impact" areas at different levels of exposure. Simple CI designs with either one or two exposure areas were applied to Dome Mine and Myra Falls Operations and to the fish monitoring tools evaluated at Mattabi Mine. The choice of design was dictated largely by the hydrological (i.e., dilution) characteristics of the site.

It was determined by the AETE Technical Committee that an appropriate sampling effort per mine site of about 20 to 25 field samples was a reasonable trade-off between feasibility and cost and statistical power and robustness (EVS *et al.,* 1997). A quality management plan was developed to ensure that the data were collected in a consistent and accurate manner among all mine sites. There was overlap in the field crews and project management for each site's activities to ensure consistency. Standard Operating Procedures (SOPs) were developed for all aspects of the field sampling program and the details of the methods for sample collection are provided in the individual site reports (BEAK, 1998a,b; BEAK and Golder, 1998b,c).

TABLE 2.1: SUMMARY OF HYPOTHESES TESTED AT EACH MINE SITE

Hypothesis	Heath Steele	Dome Mine	Mattabi Mine	Myra Falls
H1: Comparison of Sediment Toxicity Tests		X	X	X
H2: Comparison of Metals in Fish Tissues		X	X	
H3: Comparison of Metallothionein in Fish Tissues		X	X	
H4: Comparison of Metal vs Metallothionein in Fish Tissues	X	X	X	
H5: Effects on the Fish Community - CPUE	X		X	
H6: Effects on Fish or Benthic Communities	X	X	X	X
H7: Effects on Fish Growth	X	X	X	
H8: Effects on Fish Organ Size or Reproduction		X	X	
H9: Relationship between Water Chemistry and Biological Responses	X	X	X	X*
H10: Relationship between Sediment Chemistry and Biological Responses	X	X	X	X
H11: Relationship between Sediment Toxicity and Benthic Invertebrates		X	X	X
H12: Relationship between Metals or Metallothionein in Fish and Metal Concentration in Water or Sediment	X	X	X	
H13: Chronic Toxicity Linkage with Fish and Benthos Monitoring Results	X	X*		X*

X* - Because of site-specific constraints, these hypotheses were only evaluated qualitatively.

TABLE 2.2: TOOLS EVALUATED IN THE 1997 AETE FIELD STUDIES

Toolbox	**Tool**	**Method**
Water Chemistry	Total metals (ICP-MS) Dissolved metals (ICP-MS)	U.S. EPA Method No. 200.8 0.45 μm filtered
Sediment Chemistry	Total metals (ICP-MS) Partial metals (ICP-MS oxide-bound fraction) Simultaneously extracted metals (metal monosulphide fractions)	nitric acid/hydrogen peroxide extraction hydroxylamine hydrochloride cold hydrochloric acid digestion
Effluent Chronic Toxicity	*Ceriodaphnia dubia* (water flea) *Pimephales promelas* (fathead minnow) *Selenastrum capricornutum* (algae) *Lemna minor* (duckweed)	Environment Canada 1992a Environment Canada 1992b Environment Canada 1992c Saskatchewan Research Council 1995, 1996
Fish Tissues (metal analysis and metallothionein - low molecular weight metal binding proteins)	Gill Kidney Liver Muscle Viscera	filaments whole organ whole organ, gall bladder removed dorsal boneless fillet entire gut from small fish
Sediment Toxicity	*Chironomus riparius* *Hyalella azteca* *Tubifex tubifex*	Environment Canada 1997a Environment Canada 1997b ASTM E1384-94A, 1995
Fish Health Indicators	Weight Length Gonad weight Liver weight Fecundity	total body weight (grams) fork length (mm) total gonad weight (grams) total liver weight (grams), gall bladder removed total number of eggs per female
Fish Population/Community Health Indicators	CPUE BPUE	total number of fish per standardized effort electrofishing or gill netting total weight of fish per standardized effort
Benthic Community Health Indicators	Number of taxa Abundance EPT Index Indicator taxa	number of taxa at lowest practical level of taxonomy total invertebrate abundance number of genera in mayfly, caddisfly and stonefly orders abundance of indicator taxa generally at generic level

2.2 Hypothesis Testing Methods

The general reasoning behind all of the hypotheses is that a mine "effect" is a measurable difference between reference and exposure locations, and/or a trend between locations that are exposed to different degrees. The hypotheses address either the ability of a particular monitoring tool to detect such an effect (and, in aggregate, whether an effect exists) (e.g., H5 to H8), or the **relative** ability of two different monitoring tools to detect such an effect (e.g., H1 to H4). Hypotheses H9 through H12 address the **relative** ability of two monitoring tools to detect a correlation between specific predictor (exposure) and response variables (effect), whereas Hypothesis H13 addresses the ability of a particular toxicity testing tool to show such a correlation.

These different types of hypotheses required different methods of statistical analysis which are detailed in each of the site reports. The following subsections summarize the statistical approaches applied for each category. In all cases, appropriate data transformations were applied prior to statistical analysis, such as log transformation for chemical concentrations, or other parameters that span a wide range, and arcsine square-root transformations for percent response variables. Concentrations of chemical parameters that were below method detection limits were used in data analyses as half the detection limit. A significance criterion of $p \leq 0.05$ was used for all the statistical analyses and use of the term "significant" implies that this criterion was met. It should be recognized that the term "predictor" variable is not intended to mean that the measure of exposure used (e.g., metal concentration in water) can be used to "predict" a specific biological response at all mine sites or in other surveys at these mine sites. Nor does it imply that the predictor is necessarily the cause of a biological effect. Rather, the predictive ability is only suggested by correlation between effect and exposure measures.

2.2.1 H1 through H4 - Comparison of Tools as to Ability to Detect an Effect

Hypotheses H1 through H4 are tool comparison tests. Tools (response measures) were tested pairwise to determine their relative ability to detect a mine related impact. From a group of comparable tools (e.g., toxicity tests), this comparison allows the selection of the tool or tools that can best measure the impact of mine-related exposure. H1 compares toxicity endpoints (i.e., sediment toxicity compared between common test organisms), whereas H2 through H4 examine metals and metallothionein in various fish tissues. Specifically, H2 compares the concentration of a single metal at a time, between two organ tissues; therefore, tissues are the tools for comparison. Similarly, H3 compares metallothionein concentration between two organ tissues, so again tissues are the tools being compared. In H4, a metal concentration was compared to metallothionein concentration in the same organ tissue or group of tissues, so the tool comparison in this case was between metal and metallothionein, rather than between two tissues. In all four hypotheses, the method of statistical analysis was the same.

Analysis of Variance (ANOVA) was used to partition the overall variance in the response measures into various terms representing effects of particular interest (e.g., mine effect). In the case of CI designs, there was limited opportunity for partitioning of “among area” effects. In order to determine whether two monitoring tools differed in their ability to detect mine effects, simple ANOVAs were used to identify whether there was a significant area x tool interaction (i.e., two tools showing different patterns of response with exposure level). If there was, then a plot of the interaction was examined to confirm that the pattern was consistent with one tool being a better indicator of mine effects.

For the testing of hypotheses with only a single level of exposure, mine (area) effects were identified only by detection of reference-exposure differences using ANOVA. Tools were compared simply by comparison of the reference-exposure response patterns. A significant area x tool interaction suggested a greater effectiveness in the tool with the larger difference between exposure area response and reference area response.

For mine sites where gradient designs could be applied, the ANOVAs were used to compare tool effectiveness in two ways:

- by determining if the tools differed with respect to their reference-exposure difference (a larger reference-exposure difference indicates greater effectiveness); and
- by determining if the tools showed a similar linear trend or gradient in response within the exposure area (a stronger trend indicates greater effectiveness).

The ANOVA partitions overall variance in the response measure into a number of terms, representing effects of particular interest. These included:

- A “Ref vs Exp x Tool” term which indicates whether the Reference versus Exposure difference is similar for both tools (e.g., for metallothionein and copper in tissue). A larger difference between the reference and exposure means for one tool relative to the other would indicate a greater effectiveness for the tool with the greater difference.

- A "Linear Trend x Tool" term which indicates whether the linear trend in the Exposure area (e.g., from near-field to far-field) is similar for both tools.

2.2.2 H5 Through H8 - Fish CPUE, BPUE, Growth, Organ Size and Benthic Community Responses

Hypothesis H5 compares fish catch-per-unit-effort in reference and exposure areas. Hypothesis H6 compares a number of indices collected from benthic and fish communities (e.g., number of taxa, number of individuals, abundance of particular indicator taxa) in the areas compared. Hypothesis H7 examines area differences in weight and length (age adjusted if necessary), and H8 tests for area differences in liver and gonad weight for each sex and fecundity (body weight adjusted if necessary).

Hypotheses H5 through H8 address the ability of a particular population or community index tool (response measure) to show a relationship to mine exposure. For CI designs, a response variable such as fish growth or number of benthic taxa was compared by ANOVA for stations across the two or three areas (e.g., reference, near-field, and far-field) to determine whether area means were significantly different (i.e., whether the response measure varies more among areas than it does within areas). If so, data plots were examined to determine whether the pattern of area differences was consistent with a mine effect.

ANOVA was used to address these hypotheses. The ANOVA partitions the overall variance in the response measure into a number of terms representing effects of particular interest. In cases using CI designs, with only one reference area and one or two exposure areas, there is limited opportunity for partitioning of "among-area" effects. In order to determine whether an index tool can detect a mine effect, a simple test by ANOVA was used to determine whether the index varies more among areas than it does within areas. If so, then an examination of the pattern of differences between areas was undertaken to confirm that the pattern of response with exposure level was consistent with a mine effect.

In cases using gradient designs, ANOVAs were used to partition overall variance in the response measure into a number of terms, representing effects of particular interest. These included:

- An "Among Reference" term which indicates whether the various Reference reaches were similar to each other. This term was quantified in order to indicate whether reference reaches were differentially influenced by some factor (e.g., habitat) that may also be confounding effects in the exposure area.
- A "Ref vs Exp" term which indicates whether the Reference and Exposure reaches are similar to each other. A reference-exposure difference is generally indicative of tool effectiveness, assuming that the direction of the difference was consistent with impact.
- A "Linear Trend" term which indicates whether there was a linear trend in the Exposure area (e.g., from near-field to far-field). A significant linear trend, (i.e., a near-field to far-field gradient) was indicative of tool effectiveness, assuming that its direction was consistent with the expected impact.

2.2.3 H9 through H12 - Tool Integration Hypotheses

Hypotheses H9, H10 and H11 address the **relative** ability of two monitoring tools to detect a mine effect in the form of a correlation between responses measured and exposure. For example, in H9, dissolved metal in water was compared to total metal in water, for each of the key metals, to determine whether these two monitoring tools differ in their correlation with a response measure, such as number of taxa. Correlation analysis was used to address this hypothesis, as described below.

The squared coefficient of correlation (r^2) between the response measure (Y) and each predictor variable (X1 or X2) indicates the proportion of variance in the response measure that is explained by the predictor. The best predictor, for each pair compared, is the one which explains the highest proportion of variance (i.e., has the highest r^2 and hence the highest r). No statistical test was performed to determine whether r_1 differs significantly from r_2, since the two r values are based on the same Y data set and are not independent. However, the individual r values were tested for statistical significance. Two r values were compared, to draw inferences about which monitoring tool is better, only when at least one of the r values was of the correct sign (negative or positive) to suggest a mine effect, and statistically distinguishable from zero based on a one-tailed test.

In situations where a CI design with only two exposure areas was used, the degree of significance indicated by correlation analysis may be somewhat overstated, since the sampling stations are clustered in two or three areas (one reference and one or two exposure areas) and therefore may not be independent as assumed by the correlation test procedure. This situation occurred with fish at Dome and Mattabi and with benthos in Myra Creek.

For the gradient design situations at Heath Steele and Mattabi, correlation analysis was carried out wherever possible without reference site data. A correlation found using exposure site data only was considered more indicative of a mining effect than one requiring the reference site data in the data set. This is because the reference site data may tend to "drive" the correlation due to a clustering of data points at extreme low metal concentrations. For CI designs, all reference and exposure data were generally required for the analysis owing to a general lack of sufficient data for analysis at exposure sites alone (e.g., only one or two values in the "X" domain).

When differences between r values are small (e.g., ≤ 0.1), even though one or both r values may be statistically significant, a judgement is generally not made that the tool with the slightly higher r value is better able to detect an effect. Also, the correlations are generally calculated for many exposure measures (metals), so that judgements with respect to which exposure measure tool (e.g., total versus dissolved concentration in water) is more strongly correlated with biological response are made by the weight-of-evidence based on all r values for each tool. The exposure and response measures selected for inclusion in the analyses were those which showed an apparent spatial relationship to the mine site (i.e., trend among exposure reaches or difference between reference and exposure reaches).

Hypothesis H9 is tested by correlation between benthic or fish index values and metal concentrations in water (dissolved or total) from stations in areas sampled (reference, exposure areas). Hypothesis H10 is tested in a similar manner by correlation of benthic or fish index values versus sediment chemistry correlations and sediment toxicity versus sediment chemistry correlations, based on exposure and reference data. The sediment chemistry tools include total metal concentrations (hydrogen peroxide/nitric acid extraction), partial metal concentrations (hydroxylamine hydrochloride extraction) and the ratio of the molar sum of simultaneously-extracted metals (SEM) and acid volatile sulphide (AVS). Metals included in the SEM value are Cd, Cu, Ni, Pb and Zn. These are the metals normally contributing to toxicity and potentially rendered non-bioavailable by the formation of metal monosulphides.

Hypothesis H11 examines the remaining component of the "sediment quality triad" - the correlation between benthic indices and sediment toxicity - based on near-field, far-field and reference stream data. The toxicity tests include amphipod (*Hyalella azteca*), chironomid (*Chironomus riparius*) and oligochaete (*Tubifex tubifex*) tests on sediment samples from each stream station.

Hypothesis H12 examines the correlation between water and sediment chemistry measurements and concentrations of metals and metallothionein in fish tissues. For fish, station means were used as values in order to permit pairing with water and sediment chemistry values.

2.2.4 H13 - Chronic Toxicity - Linkage with Benthic and Fish Results

Hypothesis H13 addresses the ability of a particular effluent toxicity testing tool to predict a mine effect that has been otherwise demonstrated (e.g., a benthic index response to exposure). For example, H13 might address whether a specific benthic response can be predicted from effluent toxicity to *Ceriodaphnia*, *Selenastrum*, fathead minnow or duckweed.

In order to test this hypothesis, it is necessary to estimate the receiving water toxicity to each species in the near-field and far-field areas, based on the effluent toxicity information and the expected downstream dilution of effluent close to the time of the survey. This could only be done at Heath Steele. At Myra Falls and Dome Mine, this hypothesis was evaluated qualitatively because there was only one level of exposure at Myra Falls and, at Dome, the mine was not discharging effluent at the time of the survey.

Water toxicity, like effluent toxicity, can be expressed as a % inhibition (i.e., for *Ceriodaphnia* as % inhibition of reproduction). The % inhibition increases with effluent concentration. The IC25 concentration produces 25% inhibition, and the IC50 concentration produces 50% inhibition. These two concentrations, obtained from the effluent toxicity test, define the % inhibition vs concentration relationship. This relationship was used to estimate the % inhibition that would be expected at each effluent concentration that exists in downstream reaches.

The % inhibition vs concentration relationship has a sigmoid form, such that % inhibition increases most rapidly with concentration in the vicinity of the IC50 concentration. It is standard practice to transform both variables (i.e., % inhibition and effluent concentration) to make a linear relationship, in order to facilitate estimation of % inhibition at any concentration. A probit (or Z) transformation of % inhibition and a log transformation of effluent concentration will accomplish this (Finney, 1971).

Water toxicity was estimated in this manner for each exposure area downstream of the mine, based on three different effluent samples and up to four different toxicity test methods (*Ceriodaphnia*, fathead minnow, algae, duckweed). Each of these toxicity variables was tested for correlation with each of the field measurements of biological response, such as fish CPUE, and plots were produced to illustrate some of the stronger relationships.

The correlation coefficient (r) was computed and tested for significance using a one-tailed t-test. Significance depends only on the magnitude of r and a sample size (n). For n=5 exposure reaches, r must be greater than 0.81 (i.e., $r^2 > 0.65$) to produce a significant correlation ($p < 0.05$). A significant correlation (r) indicated that the toxicity tool may be useful as a predictor of the in-stream biological response measure. It did not, of course, prove that effluent was responsible for any observed pattern in biological response downstream from the mine. The toxicity test methods that generally provide the highest correlations with biological response measures are considered the best.

2.2.5 Triad Hypotheses

The "triad" hypothesis addresses the issue of whether chemical contaminants may be responsible for biological "effects" that are apparent in the study area. This hypothesis has not been articulated explicitly in the set of 13 hypotheses that were developed by the AETE (Section 1.0); however, it is consistent with the interest in H9 through H13 about the ability or relative ability of monitoring tools to detect correlations or relationships between chemical, toxicological and biological parameters. The basic approach to evaluation of the triad hypothesis is to simultaneously examine three types of correlations: chemical-toxicological (C-T), toxicological-biological (T-B) and chemical-biological (C-B). These are the three "arms" of the triad that would support an interpretation that chemical contaminants are responsible for biological effects. There should be significant correlations on all three arms before the hypothesis that chemical contaminants are the cause of the effect is accepted. None of the 13 hypotheses were specific to the testing of C-T correlations; however, it was tested as a component of the sediment quality triad.

Statistical approaches to triad evaluation follow Green and Montagna (1996) and Chapman (1996). One approach is to examine the three bivariate correlations (C-T, T-B, C-B) for different sets of chemistry, toxicity and biology monitoring tools. Then, the overall evaluation of the triad hypothesis is based on "weight-of-evidence" considerations (i.e., are there sets of parameters showing significant C-T, T-B and C-B correlations, how many sets are there that meet this criterion, and how strong are the correlations in general?). This approach is simple, but rather tedious when there are many different chemistry, toxicity and biology monitoring tools to be paired in different ways.

A more holistic approach was applied using principal components analysis (PCA) to reduce the large number of variables to one or two dominant principal components (PCs) representing the mine effect gradient in chemistry (based on the original chemical variables), one or two representing the gradient in toxicity, and one or two representing the gradient in biology. Then multiple correlation coefficients (R) were computed using the PC variables to represent the dominant C-T, T-B and C-B correlations (if any) on each arm of the triad. Mantel's test was used to produce a single measure of concordance on each arm of the triad, equivalent to R^2. Finally, Bartlett's test of sphericity was applied to determine if there is a significant overall concordance across the three arms of the triad. This approach was applied effectively using sediment chemistry, sediment toxicity and benthic invertebrate community data only, as water-based toxicity measurements were not available for receiving water samples.

2.2.6 Statistical Power

The statistical power of the study designs was evaluated using the Borenstein and Cohen (1988) computer code for power analysis. For sediment-related hypotheses, the total sampling effort for all sites was sufficient to expect that an effect size (average difference between groups) of two within-group standard deviations could be detected with a power of 0.8 or better (i.e., chance of false-negative conclusion (beta) less than 0.2) using a significance criterion based on a chance of false-positive conclusion (alpha) less than 0.05.

For fish-related hypotheses, the sampling effort at the sites was sufficient to expect that an effect size of one to two within-group standard deviations could be detected. The absolute difference indicated by the one or two standard deviations varied from one monitoring parameter (effect measure) to another and among sites.

For Hypotheses H9 to H12, the sampling effort was sufficient to detect strong chemistry-biology-toxicity correlations (those that exceed r=0.7; power=0.8).

2.3 Overall Cost Effectiveness Evaluation

The results of the hypotheses testing for each of the mine sites were summarized to determine the overall effectiveness of a monitoring tool and to assess the consistency of the tool among the different mine sites. The cost for the implementation of each tool was calculated based on the actual level of effort (hours) applied in the 1997 field evaluations and incorporating typical charge rates used by environmental consultants and laboratories in Canada. The costs assigned to various tools are only provided as guidance, because costs can vary widely among provinces, laboratories and environmental consultants. The hours of effort applied were based on biologists who generally averaged more than ten years of environmental monitoring experience. Some of the field crew leaders had 15 to 20 years of experience in conducting environmental effects monitoring studies at mines in Canada.

For each set of tool comparisons, other considerations which could have a profound effect on the overall cost of the tool were highlighted. For example, the costs to mines located in remote areas to procure dry ice for metallothionein analysis and ensure samples are frozen during transport would be much higher than the costs to a mine that is located close to a major town, where dry ice is readily available.

If the overall comparison of two tools concluded that both were considered equally effective then the least costly of the two tools was selected as the most cost-effective tool. Likewise, if the lower cost tool was also found to be the most effective tool then it was selected as the most cost effective. However, in cases where the most effective tool happened to also be more expensiv, a subjective decision had to be made as to whether or not the additional effectiveness of the tool reasonably justifies the additional cost (e.g., dependent upon the degree of greater effectiveness and magnitude of cost differences).

3.0 COST EFFECTIVENESS OF AQUATIC EFFECTS MONITORING TECHNOLOGIES

3.1 Introduction

The 1997 Field Evaluation program evaluated several aquatic effects monitoring "tools" considered by the AETE program. These tools were evaluated through testing of the thirteen hypotheses pertinent to the 1997 field program, as well as by examination of tool performance indicators other than those specific to these hypotheses (e.g., sediment quality triad).

Monitoring tools may be organized within "tool boxes" under the four guiding questions formulated under the AETE program to develop the hypotheses tested (from Section 1.1):

1. Are contaminants getting into the system?
2. Are contaminants bioavailable?
3. Is there a measurable (biological) response? and
4. Are contaminants causing the response?

Tool boxes and monitoring tools may be categorized under these four questions. Some tools may logically fit under more than one question; for example, toxicity testing tools may fit under Questions 1, 2 or 3. Table 3.1 provides a reasonable framework for organization of these tools, although alternate frameworks may be equally valid.

The fourth question cannot be answered by the application of individual tools, unlike the first three questions. Rather, the fourth question can be answered only by integrating the use of tools between and among tool boxes through testing for statistical linkages between potential cause and effect variables (e.g., do chemical concentrations and biological measurements correlate with one another?). The most effective tools are clearly those used in combinations that provide a yes answer to Question No. 4, assuming of course that the biological tools have clearly demonstrated a mine effect. The sediment quality triad provides a means of addressing Question No. 4. It should also be emphasized that a yes answer to the fourth question does not directly prove cause-effect relationships, but simply infers cause-effect by correlation analysis. Thus, when it is said that a causative agent is effective in "predicting" a biological response, this "prediction" is *a posteriori*, and is specific to the data set analyzed. Thus, a "predictive" relationship is not necessarily predictive at another site or with another data set from the same site.

TABLE 3.1: GUIDING QUESTIONS, TOOL BOXES AND TOOLS CONSIDERED IN THE 1997 FIELD PROGRAM

Question	Tool Boxes	Tools
Are contaminants getting into the system?	Water chemistry	• total metal concentrations • dissolved metal concentrations
	Sediment chemistry	• total metal concentrations • partial metal concentrations • acid volatile sulphide and simultaneously extracted metals
Are contaminants bioavailable?	Fish tissues	• organ/tissue metal concentration • organ/tissue metallothionein concentration
Is there a measurable response?	Effluent chronic toxicity[1]	• fathead minnow survival and growth test • *Ceriodaphnia dubia* (microcrustacean) survival and reproduction test • *Selenastrum capricornutum* (algae) growth test • *Lemna minor* (duckweed) growth test
	Sediment toxicity	• *Chironomus riparius* (larval insect) survival and growth test • *Hyalella azteca* (crustacean) survival and growth test • *Tubifex tubifex* (aquatic worm) survival and reproduction test
	Fish health indicators	• fish growth (length, weight and age) • fish organ size, fecundity
	Fish population/community health indicators	• fish catch-per-unit-effort (CPUE - by species and total) • fish biomass-per-unit-effort (BPUE - by species and total)
	Benthic community health indicators	• densities of benthic invertebrates • numbers of benthic taxa • benthic community indices (e.g., EPT index)
	Periphyton community health indicators	• periphyton community biomass • numbers of periphyton taxa
Are contaminants causing the response?	Pair-wise combinations of the above tool boxes	• chemistry x biology tool correlations • toxicity x biology tool correlations • chemistry x toxicity tool correlations • Sediment Quality Triad

[1] Effluent chronic toxicity measured in the laboratory may also be categorized under Questions 1 or 2 (Are contaminants getting into the system?).

The hypotheses are formulated to answer two general types of questions:

- Is the tool effective in measuring a mine effect (i.e., is there a reference - exposure difference or an exposure area trend)?; and
- Is one tool more effective than another in measuring a mine-related effect?

The "effectiveness" of monitoring tools as discussed herein is specific to the four mine sites studied. The data summarized in this document represent four mine sites considered in the AETE 1997 Field Evaluation Program, and only four of numerous mine sites across Canada. A tool that is found to be of little value at any or all of these four sites for detecting mine effects may be very useful at other sites and vice versa. Therefore, the reader is cautioned not to assume that the conclusions drawn with the data will necessarily be broadly valid at all mines across Canada. As shown in this document, monitoring tools can respond very differently from site to site. Also, the presence or absence of a particular mine-related effect may simply reflect exposure level or metal bioavailability at the site. In the latter case, the absence of an effect may simply indicate that the tool was suitable for showing no effect. However, the degree of impact found at all sites and the aqueous and sediment concentrations of metals present were consistent with conditions which should demonstrate the effectiveness of monitoring tools unless they are insensitive.

3.2 Presentation Format

For each of the four guiding questions, tables have been compiled which summarize and compare the tools' effectiveness in showing a mine-related effect or trend at each of the four mine sites. A tool is considered to have demonstrated a mine effect (table symbol - √) if it showed a significant mine -related trend or response or if it was a useful predictor for an observed mine-related effect. An asterisk (*) beside the "√" symbol identifies a tool that was considered to be more effective in demonstrating a mine effect. A tool which did not demonstrate a mine effect was identified by the table symbol "X". A tool that was considered partially effective (table symbol - "?") in demonstrating a mine effect was one that generally showed a mine-related effect or trend for a limited number of response endpoint measures. For example, if the number of benthic invertebrate taxa and total density affected by the mine and water concentrations of total copper showed a significant negative correlation with one endpoint measure but not with the other, total copper would be considered partially effective. Another example would be, if the sediment toxicity *Tubifex* test showed a significant mine-related effect with the number of cocoons produced per worm but the number of young produced per worm did not show a mine effect, then this tool would be considered partially effective. Tools that were not tested at a particular site are coded "NT" (not tested). Comments taken directly from the individual site reports are also provided for each tool to demonstrate the level of effectiveness.

3.3 Are Contaminants Getting Into the System?

3.3.1 Water Chemistry Tool Box

At all four mine sites, water chemistry sampling at the exposure stations demonstrated that metals and other contaminants (e.g., nutrients, ammonia, cyanide) were "getting into the system". This was demonstrated by elevated downstream concentrations of total and dissolved metals, as well as other contaminants. Total and dissolved fractions of the same metal were similarly effective in showing that contaminants were entering the receiving environments of the mines (Table 3.2). Generally, at all mines a high percentage of the metals was in dissolved form; consequently, there was little difference in the mine-related trends shown by total metals versus dissolved metals. The data for the Dome Mine may be influenced by the fact that the mine had stopped discharging effluent two months prior to the field survey. Despite this, elevated aqueous metals were still detected downstream of the Dome discharge, probably owing to sources in river sediments, abandoned mine wastes and/or to residual Dome mine effluent in the very low-flowing river.

The results from testing of Hypothesis H9 to show linkages between the metals (total or dissolved) entering the system and the statistically significant mine-related biological effects are provided in Table 3.3. For two of the sites (Dome Mine and Mattabi Mine), the correlations with metals were limited. In the case of Dome Mine, most of the metals that were elevated in the exposure area did not show mine-related trends with biological responses. This may be a result of the fact that the mine was not discharging at the time of the survey. At Mattabi there were few biological effects that were consistent with metal exposure so most of the correlations did not appear to be mine-related. At Dome and Mattabi mines, total and dissolved metals were effective predictors of a limited number of effects noted on fish and benthic communities. At Heath Steele there were a number of total and dissolved metals that were mine-related and correlated with the biological effects. At Myra Falls the correlations could only be assessed in a qualitative manner. Metals were higher in the exposure area and there were a number of biological measures that demonstrated a mine effect. Overall, the strength of the correlations which appeared to be mine-related were similar for total and dissolved metals, because dissolved and total metal

TABLE 3.2: TOTAL VERSUS DISSOLVED METALS AS INDICATORS OF EXPOSURE

Mine Site	Total Metal	Dissolved Metal	Comments
Dome Mine	√	√	There were increased concentrations of total and dissolved Cu, Mg, Se, Ag, Co, Ni and K at all river exposure stations. All metals detected above MDL were elevated in exposure lake. Total and dissolved metals were equally effective in demonstrating exposure.
Heath Steele	√	√	There was a gradient in total and dissolved Zn, Cd, Cu, Pb, Fe and Al in exposure area. Dissolved and total metal concentrations were similar in effectiveness as indicators of exposure.
Mattabi Mine	√	√	There were increased concentrations of total and dissolved Zn, Cu, Pb and Cd in exposure area (Bell Creek). Total and dissolved concentrations approximately equal in reflecting elevated metal concentrations.
Myra Falls	√	√	Increased concentrations of most total and dissolved metals in exposure area (Myra Creek). Total and dissolved equally effective in demonstrating exposure.

NT - Not tested.
√ - Effect demonstrated.
X - Effect not demonstrated.
? - Effect partially demonstrated.
* - More effective at demonstrating an effect.

TABLE 3.3: TOTAL VERSUS DISSOLVED METALS AS PREDICTORS OF BIOLOGICAL RESPONSES (HYPOTHESIS H9) INCLUDING METALLOTHIONEIN AND BIOACCUMULATION (HYPOTHESIS H12)

HYPOTHESIS H9

Mine Site	Total Metal	Dissolved Metal	Comments
Dome Mine	?	?	Total and dissolved arsenic negatively correlated with fecundity whereas Mg and Ni positively correlated. No mine-related correlations with benthic indices except for negative correlations of total and dissolved Co, Cu, K, Mg, Ni with % chironomids. Body weight-adjusted female gonad weight negatively correlated with Co and Cu. Dissolved and total metals equally effective, although limited. Correlations may be limited due to the fact the mine only discharges sporadically and was not discharging at the time of the survey.
Heath Steele	√	√	Numbers of total benthic taxa and EPT taxa are reduced and dominance of tolerant chironomids increases with increasing metal in water. Fish CPUE, BPUE and number of fish taxa decrease with increasing metal in water. Total and dissolved metals (Zn, Cd, Cu, Pb, Al) correlated with biological effects. Strength of the relationships similar for dissolved and total metals versus biological responses.
Mattabi Mine	?	?	Similar negative correlations for a limited number of dissolved and total metals (Cu, Fe, Mg, Zn). Pb concentrations of both appeared unrelated to most biological effects. H9 to be interpreted with caution due to study design limitations (CI design).
Myra Falls	√	√	Only one exposure area; therefore,correlations not possible. Dissolved and total metals higher in exposure area where effects on benthos were observed. Dissolved metals were a high percentage of total metal; therefore, correlations with benthic effects would have been similar. H9 assessed qualitatively.

HYPOTHESIS H12

Mine Site	Total Metal	Dissolved Metal	Comments
Dome Mine	√	√	Total and dissolved Co, Cu, Ni correlated with pearl dace viscera. No mine-related response between MT and aqueous metals. No overall difference in strength of the correlations of total and dissolved aqueous metals with viscera concentrations.
Heath Steele	√	√	MT in wild and caged salmon viscera correlated with metals in water; metals in salmon did not correlate with metals in water. MT in blacknose dace viscera correlated with Pb in water. Strength of the correlations of total and dissolved and metals and MT in tissues similar.
Mattabi Mine	?	?	In sucker, only Pb and Zn in gill were correlated with aqueous metal concentrations. MT concentrations in sucker tissues were unrelated to exposure concentrations. In pike, Pb in kidney and gill, and Zn in muscle, were correlated with aqueous concentrations of the same metals. Pike MT levels in kidney and gill were correlated with Cd, Pb and Zn in water. These metals in tissue showed similar correlations with corresponding metals in water.
Myra Falls	NT	NT	

NT - Not tested.
√ - Effect demonstrated.
X - Effect not demonstrated.
? - Effect partially demonstrated.
* - More effective at demonstrating an effect.

Note: Dissolved metal concentrations represented a high percentage of total metals concentrations for most key metals at all sites. Thus, correlations between biological responses and aqueous metal concentrations were similar for total and dissolved metals.

concentrations tended to co-vary, and dissolved metal levels represented a high percentage of total metal levels for most metal parameters.

In testing of Hypothesis H12 to show linkages between dissolved and total metals in water and bioaccumulated tissue metals or metallothionein response, both dissolved and total metals showed some significant correlations. At Heath Steele, both dissolved and total concentrations of various metals were significantly correlated with tissue metals and metallothionein. At Dome, the correlations were only significant with tissue metals and not metallothionein, because metallothionein did not show a mine-related trend. At Mattabi, the correlations of aqueous metals and metallothionein were only seen with pike and not with white sucker. For the reasons noted above, the strengths of the correlations between dissolved or total metals versus either tissue metal or metallothionein response were similar.

None of the AETE sites offered conditions where total and dissolved metal concentrations differed substantially from each other for key metals. The relative effectiveness of total versus dissolved metal concentrations in producing biological effects could be more appropriately assessed at a mine site where dissolved and total metals differed substantially, perhaps where total suspended solids concentrations are high in the water column.

Costs

Water sample collection costs for total or dissolved metals are the same except for the additional time required to filter the samples for dissolved metals. The time required for filtration could potentially be reduced if a more efficient protocol than used here is developed for field filtration. Analytical costs are the same for both types of sample, although there are some additional QA/QC costs for checking the filtering process. There are additional costs for disposables (e.g., extra bottles, syringes and filters) for the dissolved metal samples. Table 3.4 compares the unit costs between the total and dissolved metal tools. For dissolved metal analysis there is approximately a 38% increase in the unit cost when compared to total metals.

Other Considerations

The syringe used here and recommended by chemists with the Geological Survey of Canada (GSC) was difficult to procure in Canada. In this case, importation of the syringes from the U.S. required over one month. However, in the future, it may be possible to identify Canadian

TABLE 3.4: COST COMPARISON BETWEEN TOTAL AND DISSOLVED METALS

Cost Items	Tool	
	Total Metal	Dissolved Metal
Filtering	-	$12.00*
Analytical (ICP-MS)	$65.00	$65.00
Disposables	-	$5.00
Additional QA/QC (filter blanks)	-	$8.00
TOTAL UNIT COST	$65.00	$90.00

* Filtering time of 15 minutes per sample at a technician rate of $50.00 per hour. This time could potentially be reduced if a more efficient protocol is developed for field filtration in the future.

distributors of similar products. The purchase of similar filtration materials for ultra-trace metal work will require careful planning in future field studies requiring dissolved metal determinations.

The commercial laboratory used required very specific instruction to provide materials consistent with the recommendations provided by GSC. For example, commercial laboratories often provide low density rather than high density polyethylene containers for metal samples, and may also provide containers with coloured lids such as “Falcon” tubes to consultants or mining companies. GSC has shown that such containers can contribute low levels of metals to water samples, and thus may not be suitable in aquatic effects monitoring where metal concentrations of interest are equal to or often below surface water quality guidelines.

The filtration procedure involved squeezing the water through a syringe-mounted filter, and was somewhat difficult and time-consuming due to the slow rate of filtration, rinsing requirements, etc. Also, where suspended solids levels were higher, filters became quickly clogged and required replacement.

Sample contamination was apparent in the dissolved metal results for one of the sites (Myra Falls). Because of the additional sample handling and materials to filter samples, the risk of sample contamination is substantially increased when compared to water sample collection for total metals.

Conclusion

Overall, in comparing the effectiveness of dissolved and total metals to indicate metal exposure and to correlate to the measured mine-related biological effects, both tools were equally effective in predicting mine effects on biological communities and bioaccumulated metals or metallothionein response. This coupled with the fact that dissolved metals are more expensive and have an increased risk of sample contamination supports the higher priority use of total metals in routine environmental effects monitoring. Dissolved metals are sometimes considered to better represent the bioavailable metal and would therefore be better predictors of effects on biological communities however, this was not supported by the data from the 1997 field evaluations. In addition, for the majority of metals, federal and provincial water and effluent quality guidelines and regulations are based usually on total metal concentrations. Therefore, in routine EEM studies, total metals would be required if a mine wanted to compare their receiving water and/or effluent quality to existing guidelines.

3.3.2 Sediment Chemistry Tool Box

Sediment concentrations of most key metals (e.g., As, Cd, Cu, Ni, Zn, Pb, Hg) demonstrated that contaminants were "getting into the system" at all mine sites. At all four mine sites, sediment chemistry sampling (periphyton used as a surrogate at Heath Steele) at the exposure stations demonstrated that metals were elevated in concentration. Although not specifically tested as a hypothesis, total metals were more effective than partial metals in demonstrating mine-related trends in sediment chemistry (Table 3.5) and demonstrating exposure of benthos and fish at all four sites. SEM/AVS ratios were only effective at showing mine-related trends at the Myra Falls site. However, at all sites the SEM/AVS ratio was not an effective predictor of acute toxicity.

In testing of Hypothesis H10 to show linkages between sediment chemistry tools and biological effects, total and partial metals were considered to be equally effective, whereas SEM/AVS ratios were not correlated with biological effects (including sediment toxicity) at all sites (Table 3.6). At Dome Mine, total and partial metals were only considered to be partially effective because a number of significant correlations were noted with the sediment toxicity results but only few mine-related correlations were seen with the benthic community effects. Influences of habitat factors (i.e., beaver ponding at some of the reference stations) and other sources of contamination may have affected the outcome of the hypothesis testing at the Dome site.

At all three sites, significant toxicity (mortality) occurred in some sediment toxicity tests at SEM/AVS ratios below 1.0, which is contrary to the SEM/AVS sediment toxicity model. In general, SEM/AVS ratios <1 may reflect non-toxic sediment conditions because some of the key metals (e.g., Ni, Pb, Cu, Cd, Zn) which are often associated with sediment toxicity will be in sulphide forms which reduces their bioavailability. However, it is possible that sediments with SEM/AVS ratios <1 will still be toxic due to the presence of other metals (e.g., arsenic, mercury) which are not included in the SEM analysis.

SEM/AVS ratios >1 often reflect sediments that may be toxic because there is insufficient sulphide to react with the bioavailable metals to make them less toxic. Again, SEM/AVS ratios >1 do not always accurately predict that sediments will be toxic because other factors, such as organic material or clay, will also bind metals, thereby reducing their toxicity.

The SEM/AVS ratio was developed to predict acute sediment toxicity and not necessarily for predicting chronic effects, including effects on the benthic community. However, it is not unreasonable to expect that, if sediments are acutely toxic, there would be some change in the

TABLE 3.5: COMPARISONS OF SEDIMENT CHEMISTRY TOOLS (TOTAL AND PARTIAL METALS, AND SEM/AVS RATIO) AS INDICATORS OF EXPOSURE

Mine Site	Sediment Chemistry Tool: Total Metal	Partial Metal	SEM/AVS Ratio	Comments
Dome Mine	√*	√	X	Mine-related trends in total Cu, Ni, Co, Fe, Mg, Ag and partial Ni, Cr, Co, Cu, Fe and Mo. Trend stronger in total metals. SEM/AVS ratio did not show a mine-related trend.
Heath Steele	√	NT	NT	Periphyton used as a surrogate for sediments. Cu, Cd, Pb and Zn all showed reference-exposure area differences. Exposure gradient seen for Pb and Fe.
Mattabi Mine	√	?	X	All metals higher in exposure area. Gradients in exposure area evident, particularly for total Zn, Cu, Cd and Pb, but were weaker for partial metals and not evident for some partial metals. No clear difference in SEM/AVS ratios between reference and exposure areas.
Myra Falls	√*	√	√	Most metals higher in exposure area. Gradient in exposure area evident for total Zn, Cu, Cd, Mo, Ag and As, but only for Cd, Cu and Zn for partial metals. SEM/AVS ratios were higher in exposure area than reference, and showed mine-related trend.

NT - Not tested.
√ - Effect demonstrated.
X - Effect not demonstrated.
? - Effect partially demonstrated.
* - More effective at demonstrating an effect.

TABLE 3.6: RELATIONSHIP BETWEEN SEDIMENT CHEMISTRY TOOLS AND BIOLOGICAL RESPONSES (HYPOTHESES H10 AND H12)

HYPOTHESIS H10

Mine Site	Sediment Chemistry Tool			Comments
	Total Metal	Partial Metal	SEM/ AVS	
Dome Mine	?	?	X	Total and partial metals similarly correlated with benthic indices and sediment toxicity. Only a few metals correlated with benthic effects. *Hyalella* mortality positively correlated with total and partial As, Co, Cr, Cu, Fe, Hg, Mg and Ni. No correlation with SEM/AVS and benthos or toxicity results. SEM/AVS ratio was not a good predictor of acute sediment toxicity.
Heath Steele	√	NT	NT	Numbers of benthic taxa and EPT taxa are reduced and dominance of tolerant chironomids increases with increasing metals in periphyton. Fish CPUE, BPUE and number of taxa decrease with increasing metals in periphyton. For most fish and benthic indices, relationships were stronger with metals in water than metals in periphyton.
Mattabi Mine	√	√	X	Correlations significant for total and partial metals (As, Cd, Cu, Fe, Pb, Ni, Se, Zn) with benthic community effects and sediment toxicity results (*Chironomus* and *Tubifex* sublethal endpoints). More total metals than partial metals were correlated in a direction consistent with mine effects. No correlation of SEM/AVS ratios with benthos or toxicity results.
Myra Falls	√	√	X	Benthic indicators and sediment toxicity were correlated with both total and partial metals for As, Cd, Cu, Zn. Correlation coefficients for total and partial metals were similar for benthic indicators. Total metals were better correlated with toxicity than partial metals. The SEM/AVS ratio did not correlate with either benthic indicators or with sediment toxicity and was not an effective predictor of acute toxicity.

HYPOTHESIS H12

Mine Site	Sediment Chemistry Tool			Comments
	Total Metal	Partial Metal	SEM/ AVS	
Dome Mine	?	?	X	No mine-related response between MT and sediment metals. Sediment total and partial Ni and Co correlated with viscera metals, also partial arsenic. No overall difference in correlations of total and partial sediment metal versus viscera concentrations, although the number of significant mine-related correlations was few.
Heath Steele	?	NT	NT	Metals in salmon did not increase with metals in periphyton. Zn, Cu and Pb in dace viscera correlated with metals in periphyton.
Mattabi Mine	NT	NT	NT	
Myra Falls	NT	NT	NT	

NT - Not tested.
√ - Effect demonstrated.
X - Effect not demonstrated.
? - Effect partially demonstrated.
* - More effective at demonstrating an effect.

benthic community structure that reflects this toxicity. Therefore, there may be a correlation between SEM/AVS ratios >1 and effects observed on benthic communities. This correlation was investigated as part of the 1997 field evaluations. However, it was found that at all sites where SEM/AVS was measured this tool was not correlated with biological effects, and was unreliable as a predictor of acute sediment toxicity.

At Dome total and partial metals were partially effective at predicting a response in tissue bioaccumulation of metals (Hypothesis H12). This was only observed for two metals of the many evaluated. At Heath Steele total metals in periphyton were only considered partially effective because correlations were found for only one of the sentinel species (i.e., blacknose dace).

Costs

Sediment sample collection costs for total and partial metals or for SEM/AVS are the same except for additional costs that may be required at some remote mine sites to keep the partial metal samples frozen. Additional costs for freezing and shipping of partial metal samples are insignificant when compared to the costs for handling samples for total metals. Analytical costs are slightly higher for partial metals compared to the unit costs for total metals and both are substantially lower than the unit costs for SEM/AVS analysis (Table 3.7).

Other Considerations

The use of partial metals requires that the field crew has access to a freezer or dry ice because the samples required freezing after collection and during transport to the analytical laboratory. In some field situations, this could increase the cost of sample collection, further decreasing the cost-effectiveness of this tool when compared to sampling for total metals.

Sediment metal analyses may be more effective than aqueous metal analyses in situations where aqueous metal concentrations are affected only sporadically (e.g., only in response to runoff or to intermittent effluent discharge), with concentrations approaching reference conditions between these impact events. This is because sediments will act to integrate metal loadings gradually over time whereas the water column may flush more rapidly. In fact, hypothesis testing showed this to be the case at Dome. Sediment metals were more highly correlated than aqueous metals with benthic parameters and viscera metals in pearl dace.

Conclusions

SEM/AVS analysis of sediment was ineffective as a predictor of biological effects, including acute and chronic sediment toxicity, at all mine sites. The costs for this analysis are substantially higher than for total or partial metals. Consequently, results from the 1997 field program do not support routine use of this tool in EEM studies at mine sites. The tool may be useful in studies designed to determine cause-effect relationships because it may assist in identifying metals that are associated with the effects.

The rationale behind the use of partial metals is that they may better reflect the bioavailability of metals and as such be better predictors of biological responses. Overall, the 1997 Field Evaluations showed that partial and total metals appeared to be equally effective as predictors of exposure to mine-related contaminants and as predictors of biological responses. However, in some cases, total metals were more effective because partial metals concentrations were substantially lower and in some cases below detection limits. Although both tools helped to show mine-related trends, correlations with total metal and biological responses tended to be stronger at some sites. Since the unit costs are lower for total metals and the fact that handling of partial metal samples may pose logistical problems at some mines, the use of total metal analyses appears to be the more cost-effective for routine use in EEM programs. The same conclusion was reached in the 1995 Pilot Study (BEAK, 1996) which stated that full extraction of metals (i.e., total metals) was better able to detect significant differences with greater power among reference, near-field and far-field areas compared to partial extraction of metals.

Another factor that would support priority use of total metals over partial metals is that provincial and federal sediment quality guidelines are based on total metal concentrations and mines are typically interested in comparing exposure concentrations to these benchmarks in EEM studies.

TABLE 3.7: COST COMPARISON OF SEDIMENT CHEMISTRY TOOLS

Tool	Unit Cost[1]
Total Metals	$80.00
Partial Metals	$90.00
AVS/SEM	$320.00

[1] Done by ICP-MS.

3.4 Are Contaminants Bioavailable?

This question is answered through the measurement of metal bioaccumulation or biochemical responses to metal bioaccumulation (e.g., metallothionein).

3.4.1 Tissue Metal Concentrations

The effectiveness of tissue metal concentrations as indicators of metal bioaccumulation is measured from the identification of differences between exposure and reference areas, with higher values in the exposure area required to indicate effectiveness. Tissues showing greater reference-exposure differences are considered more effective than those showing smaller differences for the same metal. Tissues which showed reference-exposure area differences in only one of the species studied or which varied in their effectiveness between sexes were considered to be partially effective.

At Dome Mine, four of the five tissues (kidney, liver, muscle, viscera, not gill) were effective in showing exposure-reference differences for some metals (Table 3.8). However, muscle tissue in yellow perch and viscera in pearl dace were the most effective in showing significant mine-related trends. In yellow perch the muscle tissue had much lower concentration of metals than the other tissues but was still more effective in demonstrating significant mine-related differences between reference and exposure areas. At Heath Steele, whole viscera (the principal tissue analyzed) was effective in showing mine exposure but only in one of the species studied (i.e., shown in blacknose dace but not in Atlantic salmon). Gill in caged Atlantic salmon was also effective but the same response was not seen in wild Atlantic salmon. At Mattabi Mines, gill was effective in showing a mine exposure response in both northern pike and white sucker, whereas other tissues were only partially effective and often varied in effectiveness between both species and sexes.

Hypothesis 12, which compares correlations between metals in water and metals in fish tissues (Table 3.3), indicated significant correlations between the metals entering the system from the mines and concentrations in tissues. The same result was found for metals in the viscera and sediment metals (Table 3.6).

Costs

The cost to collect fish muscle is the least expensive, although to collect liver tissue or viscera requires only slightly more effort. The collection of kidney and gill are the most time consuming because the gill filaments have to be removed from the gill rakers and all the viscera has to be

TABLE 3.8: COMPARISON OF METALS IN FISH TISSUES (HYPOTHESIS H2)

Mine Site	Tissue Type: Gill	Liver	Kidney	Muscle	Viscera	Comments
Dome Mine	X	?	?	√	√	Muscle was the most effective tissue in yellow perch showing mine-related trends in Zn, Ag, Co, Cu, Fe, Se, Al and Va. Perch liver showed some differences but only in Mo and Ni. Perch gill was unresponsive to mine exposure and kidney was only effective for Ni. Pearl dace viscera demonstrated a mine-related response in Ag, Cd, Cu, Se, Mo, Ni and Al. Caged yellow perch viscera was not effective in demonstrating mine-related exposure.
Heath Steele	?	NT	NT	NT	?	Viscera metal levels responded to metal exposure (increase in the exposure area fish) but little or no trend was present in exposure area. Viscera metals in caged salmon did not respond to metal gradient. Gill metals responded effectively to exposure over nine days in caged salmon, especially for Cd, but spatial trends were not well developed for other metals.
Mattabi Mine	√	?	?	?	NT	Muscle in white sucker was unresponsive to mine exposure except for Se but was effective in pike for Cd, Zn and Se. White sucker and pike liver were unresponsive except for Se. White sucker gill showed mine-related trend for Pb and Zn; and Pb, Co and Se in pike gill. White sucker kidney showed a mine-related effect for Se whereas, in pike, it was seen for Pb, Co and Se.
Myra Falls	NT	NT	NT	NT	NT	

NT - Not tested.
√ - Effect demonstrated.
X - Effect not demonstrated.
? - Effect partially demonstrated.
* - More effective at demonstrating an effect.

removed from the body cavity to get at the kidney. The cost for removal of the tissue and analytical costs are provided in Table 3.9.

Other Considerations

From a practical standpoint, collection of tissues for metal analysis was not problematic, although more effort was required for dissection of individual internal tissues than was necessary for viscera or for collection of muscle tissue.

At Heath Steele and Dome Mine, the metals in viscera of blacknose dace and pearl dace, respectively, were correlated with metals in periphyton and sediment, respectively. At Heath Steele, periphyton, which is a food source for blacknose dace was used as a substitute for sediments. Pearl dace are also feeders on benthos and algae and may incidentally ingest contaminated sediment. Caged fish at Dome Mine demonstrated that metal levels in the viscera were substantially higher in pre-exposure fish compared to the viscera metal concentrations after ten days of caging. These observations suggest that the viscera metal concentrations may be influenced by the contents of the alimentary canal and are not solely reflective of bioaccumulated metals. Therefore, use of viscera should include either a depuration phase to clear gut contents or physical removal of the gut contents. Either gut clearing method would increase the costs for collection of viscera.

Conclusions

The data were too variable to form any conclusions on cost effectiveness of various tissues. Viscera worked well at Dome Mine but was only partially effective at Heath Steele. The fish caging data also provided some evidence to suggest that the viscera metals may be strongly influenced by the gut content which does not reflect bioaccumulated metals. Muscle tissue had much lower metal concentrations but worked well at Dome in showing mine-related trends; however, it was only partially effective at Mattabi. Gill was an effective tissue at Mattabi but showed no mine-related response at Dome.

Typically, mining studies conducted in Canada currently use liver and muscle for monitoring of aquatic systems. The muscle tissue provides a link to human health through the use of the fish and the liver tissue provides higher concentrations of contaminants with which to monitor temporal changes in water quality. Since both the muscle and liver tissue were either effective or partially

TABLE 3.9: COST COMPARISON FOR FISH TISSUES

Cost Items	Tool				
	Gill	Liver	Kidney	Muscle	Viscera
Collection[1]	$15.00	$10.00	$15.00	$5.00	$7.00
Analytical (ICP-MS)	$80.00	$80.00	$80.00	$80.00	$80.00
TOTAL UNIT COST	$95.00	$90.00	$95.00	$85.00	$87.00

[1] Assumes two technicians, one at $50 per hour and one at $40 per hour.

effective, it is recommended that these tissues be used instead of gill and kidney. Muscle and liver are also slightly less costly to collect.

3.4.2 Tissue Metallothionein Concentrations

The effectiveness of tissue metallothionein (MT) concentrations as indicators of exposure to bioavailable metals from mines is measured by identification of differences between exposure and reference areas, with higher values in the exposure area required to indicate effectiveness. Where more than one tissue type (gill, kidney, liver) shows a significantly elevated exposure area response, the tissue(s) having larger exposure-reference differences are identified as more effective.

At Dome Mine, there were no significant reference-exposure differences in metallothionein concentrations that were related to mine exposure (Table 3.10). Metallothionein was significantly higher in reference gill and kidney, and equal in reference and near-field viscera and liver. At Heath Steele and Mattabi Mines, metallothionein was partially effective in showing a mine-related response (Table 3.10). At Heath Steele, the metallothionein response was noted in viscera of blacknose dace and caged juvenile Atlantic salmon but not in wild juvenile Atlantic salmon. At Mattabi Mines, there was a similar result, whereby a response was seen in northern pike gill and kidney but not in liver or in any of the white sucker tissues. In contrast to the 1997 studies, the 1995 Pilot Study (BEAK, 1996) showed that liver was the most effective tissue for measuring metallothionein, whereas liver was not effective in reflecting mine exposure at any of the mines studied in 1997.

The strongest mine-related metallothionein response was noted in caged salmon at Heath Steele. Interestingly, at Dome Mine, although the viscera metals decreased significantly compared to pre-exposure levels, metallothionein levels increased substantially at all cage sites, including the reference sites where water concentrations of metals were quite low. It is not known whether or not the metallothionein response is related to the stress of the caging.

Costs

There are probably only a couple of commercial labs in Canada that provide metallothionein analysis. However, BEAK has had this analysis done in the past by a university lab. Based on the price that BEAK paid for the analysis and the ratio between university prices and commercial rates for other lab analyses, it is predicted that the unit cost for metallothionein analysis would be approximately $50 to $60 per sample. There is no difference in the cost of measuring metallothionein in different tissues; however, the cost of dry ice and shipping will increase the unit costs to at least $85 assuming a seven day field trip and the collection of approximately 40 samples. Costs can vary substantially based on the availability of dry ice and airlines that will ship appropriate quantities.

Other Considerations

The collection of tissues for metallothionein analysis was not problematic at any site, although the effort required for individual organ collection was greater than for fish viscera. Maintenance of a dry ice supply was expensive at all mines and shipping by air was often problematic. There are restrictions on the amount of dry ice that can be shipped and if any live animals are on board, the airlines can not transport dry ice. Extreme diligence was required to ensure samples remained frozen on dry ice at all times and to maintain an adequate supply. It would be valuable to compare metallothionein results in tissues frozen with dry ice against those frozen at lower temperature to confirm the requirement for dry ice in the field.

Conclusions

The data appear to be too sporadic to form any sound conclusions. Based on the data from all of the mines surveyed, including those in the 1995 AETE Pilot Study, metallothionein was only partially effective at Mattabi and Heath Steele. The metallothionein responses appeared to vary among species and in the case of viscera measurements in caged fish at Dome, the metallothionein levels increased as the viscera metals decreased. It appears, from the data collected, that metallothionein measurements would be limited in effectiveness in routine EEM programs and as discussed in the following section tissue metals appear to be more cost effective in demonstrating mine exposure.

In comparing all of the data, including those from the 1995 Pilot Study, it cannot be concluded which tissue would be the most effective in monitoring metallothionein response. Metallothionein concentrations varied substantially among tissues, species and mine sites. For example, at Mattabi Mines, northern pike tissues were more effective than white sucker tissues, whereas in Rivière Bousquet, which was the exposure site for the 1995 Pilot Study, the reverse was found for these same species.

TABLE 3.10: COMPARISON OF METALLOTHIONEIN IN FISH TISSUES (HYPOTHESIS H3)

Mine Site	Tissue Type					Comments
	Gill	Liver	Kidney	Muscle	Viscera	
Dome Mine	X	X	X	NT	X	No mine-related pattern. MT in yellow perch gill and kidney higher in reference area. No significant reference/exposure difference in liver. No mine-related pattern for MT in pearl dace or caged yellow perch viscera.
Heath Steele	?	NT	NT	NT	?	Visceral MT levels responded to metal exposure (increase in exposed fish) in blacknose dace. Visceral MT showed clear spatial trend in exposed caged salmon, but no significant response seen in wild salmon. Gill MT showed clear spatial trend in exposed juvenile salmon (caged).
Mattabi Mine	?	X	?	NT	NT	MT response only significant for pike gill and kidney. These tissues were equally effective at demonstrating a mine effect. No mine-related response in white sucker tissues.
Myra Falls	NT	NT	NT	NT	NT	

NT - Not tested.
√ - Effect demonstrated.
X - Effect not demonstrated.
? - Effect partially demonstrated.
* - More effective at demonstrating an effect.

3.4.3 Cost Effectiveness Comparison between Tissue Metals and Metallothionein

The overall costs for tissue metals and metallothionein are expected to be similar based on an estimate that commercial labs will charge about $60 per sample (Table 3.11). Tissue samples for metallothionein must be in contact with dry ice at all times and this will result in logistical problems at many sites. There is always difficulty in getting sufficient quantities of dry ice on a regular basis and shipping of dry ice by air can be problematic. Shipping of metallothionein samples is also more expensive compared to the cost for shipping samples for metal analysis because the metallothionein samples have to be shipped on the next available flight to ensure that the supply of dry ice does not sublime before the samples arrive at the lab. If the samples thaw out this would require re-sampling of tissues which would not be the case for tissue metals.

Other Considerations

Collection of tissues for metallothionein results in more of an impact on the fisheries resources compared with the collection of tissues for metal analysis. The most common form of fish capture in environmental effects monitoring studies is gill netting. Inevitably, some fish die during capture, however, dead fish may still be used for metal analysis but not for metallothionein. There are probably only a couple of commercial labs in Canada which provide metallothionein analysis. If there is a demand for this analysis, it will be some time before labs are routinely providing this analysis with appropriate QA/QC.

Conclusions

Overall, the 1997 Field Studies indicated that metals in various tissues were more effective in showing a response to mine exposure than metallothionein. The tissue metals also have the added advantage of pointing to specific metals of concern and identifying those that are bioavailable. The unit cost for each sample type is expected to be similar. Therefore, tissues metal analysis appears to be a more cost-effective tool for routine environmental effects monitoring studies. The 1995 Pilot Field studies also concluded that tissue metals were more cost-effective (BEAK, 1996).

TABLE 3.11: UNIT COST COMPARISON FOR FISH TISSUE METALS VERSUS METALLOTHIONEIN

Cost Items	Tool	
	Tissue Metals	Tissue Metallothionein
Analytical	$80.00	$60.00
Dry Ice (cost and procurement time)[1]	-	$15.00
Shipping Costs[1]	$3.00	$6.00
TOTAL UNIT COST	$83.00	$81.00

[1] Based on the assumption of a one-week field campaign and collection of 40 samples.

3.5 Is There a Measurable Effect?

The answer to this question is evaluated through Hypotheses H1, and H6 through H12. The hypotheses tested were based on a measurable effect in fish or benthos (H6 through H8) and on the integration of tool hypotheses (H9 through H12) which look for correlations between the measurable effects and the potential causal agents.

3.5.1 Sediment Toxicity

The effectiveness of sediment toxicity as an indicator of metal bioavailability is measured from the identification of differences in toxicity between reference and exposure areas and/or the occurrence of trends within the exposure areas (near-field to far-field). Effectiveness is also determined by the strength of correlations between possible causal agents (metals in sediment) and sediment toxicity and between sediment toxicity and benthic community effects.

Sediment toxicity reflecting mine exposure was evident in mortality and growth impairment in *Hyalella* at Dome Mine and Myra Falls but not at Mattabi (Table 3.12). *Chironomus riparius* tests were only effective at Myra Falls. At Myra Falls, *Hyalella* and *Chironomus* tests were found to be equally effective in showing a mine-related effect. The *Tubifex* test was found to be only effective at Myra Falls and ineffective at Dome Mine and Mattabi Mines in showing a response to exposure. At Myra Falls, the magnitude of the reference-exposure difference was greatest for *Hyalella* and *Chironomus*, indicating that these tests were more effective than the *Tubifex* test.

The sediment toxicity was generally correlated with a number of sediment metals and with benthic community metrics (Tables 3.13 and 3.14, respectively). However, at Mattabi, the toxicity tests did not show a mine effect although there was toxicity at a number of stations, and only the *Tubifex* test was correlated with benthic measures. Although sediment contamination was quite severe at Mattabi, the lack of a mine-related trend in toxicity suggests that the metals are limited in their bioavailability.

Overall, sediment toxicity was effective in responding to sediment contamination and predicting benthic community responses but there was variability among sites as to which tests were most effective.

Costs

The commercial costs for *Hyalella* and *Chironomus* testing are generally similar since culturing, test monitoring and endpoint measures require similar effort. *Tubifex* tests are not widely available commercially, however the costs are expected to exceed those for *Hyalella* and *Chironomus* because the duration of the test is three times as long and greater effort is required to measure the endpoints. Table 3.15 provides a comparison of typical unit costs for each test.

Other Considerations

From a practical standpoint, sediment toxicity was readily assessed at all sites. *Tubifex* testing is not currently widely available from commercial laboratories in Canada and it will take some time for commercial labs to develop the expertise and meet QA/QC requirements. Commercial testing capability is widely available for sediment testing with *Chironomus* and *Hyalella*.

Conclusions

The *Hyalella* sediment toxicity test was the most effective at the Dome and Myra sites and correlated well with benthic invertebrate community responses and sediment chemistry. The *Chironomus* and *Tubifex* tests were less effective at showing a mine-related response and in predicting effects on benthic invertebrate communities. The cost for the *Hyalella* test is similar to the cost of the *Chironomus* test and expected to be lower than the cost of the *Tubifex* test. The *Hyalella* test was also found to be the most effective test in the 1995 Pilot Study (BEAK, 1996). The *Hyalella* tests overall appeared to be the most cost-effective sediment toxicity testing tool.

Based on the data from the 1997 Field Studies and the 1995 Pilot Study, there appears to be little indication of value added by completion of a multi-species battery of sediment toxicity tests. At Dome and Myra Falls, and at the mine studied in the 1995 Pilot Study, the results showed that *Hyalella* tests were the most effective at showing a mine-related response. The test results also correlated well with the benthic community effects that were noted at the sites. Little additional information on sediment toxicity was gained by doing the *Tubifex* and *Chironomus* tests. If it were desirable to do multi-species tests then *Chironomus* would be the next most cost-effective test and it is currently available at commercial toxicity labs. However, the *Chironomus* test was only evaluated in 1997 and, with these limited data, the effectiveness of this test could not be fully evaluated. Typically, when multi-species tests are used, they involve species from different trophic levels (e.g., effluent toxicity tests use fish, invertebrates, algae). In the case of sediment toxicity,

TABLE 3.12: COMPARISON OF THE EFFECTIVENESS OF *HYALELLA AZTECA*, *CHIRONOMUS RIPARIUS* AND *TUBIFEX TUBIFEX* TOXICITY MONITORING TOOLS (HYPOTHESIS H1)

Mine Site	Sediment Toxicity Tools			Comments
	Hyalella azteca	*Chironomus riparius*	*Tubifex tubifex*	
Dome Mine	√	X	X	No mine-related response in *Tubifex* or *Chironomus.* Mine-related trend in *Hyalella* mortality and growth.
Heath Steele	NT	NT	NT	
Mattabi Mine	X	X	X	All tests showed no significant reference-exposure differences or trends in the exposure area. Thus, no discernible difference in effectiveness. Area effects but unrelated to exposure were evident for sublethal responses in *Chironomus* and *Tubifex*.
Myra Falls	√*	√*	√	Mortality increased with exposure for *Hyalella* and *Chironomus* tests, but not for *Tubifex. Tubifex* responded in terms of reproductive effects.

NT - Not tested.
√ - Effect demonstrated.
X - Effect not demonstrated.
? - Effect partially demonstrated.
* - More effective at demonstrating an effect.

TABLE 3.13: COMPARISON OF THE RELATIONSHIPS BETWEEN SEDIMENT CHEMISTRY AND SEDIMENT TOXICITY RESULTS (HYPOTHESIS H10)

Mine Site	Sediment Toxicity Tools			Comments
	Hyalella azteca	*Chironomus riparius*	*Tubifex tubifex*	
Dome Mine	√	NT	NT	*Hyalella* mortality positively correlated with total and partial As, Co, Cr, Cu, Fe, Hg, Mg and Ni. *Chironomus* and *Tubifex* were not tested because they showed no mine response.
Heath Steele	NT	NT	NT	
Mattabi Mine	NT	√	√	Similar correlations for total and partial metals with sediment toxicity results (*Chironomus* and *Tubifex* sublethal endpoints) (As, Cd, Cu, Se, Zn for *Tubifex* only). Total metals slightly better correlated than partial metals. *Hyalella azteca* not tested because it showed no mine-related or area effects.
Myra Falls	√	√	√	Sediment toxicity was correlated with both total and partial metals for As, Cd, Cu, Zn. Total metals were better correlated with toxicity than partial metals overall.

NT - Not tested.
√ - Effect demonstrated.
X - Effect not demonstrated.
? - Effect partially demonstrated.
* - More effective at demonstrating an effect.

TABLE 3.14: COMPARISON OF THE RELATIONSHIPS BETWEEN SEDIMENT TOXICITY TOOLS AND BENTHIC INVERTEBRATE COMMUNITIES (HYPOTHESIS H11)

Mine Site	Sediment Toxicity Tools			Comments
	Hyalella azteca	*Chironomus riparius*	*Tubifex tubifex*	
Dome Mine	√	X	X	*Hyalella* mortality and growth correlated with most benthic indices. *Hyalella* test effective in predicting impacts on benthic community.
Heath Steele	NT	NT	NT	
Mattabi Mine	X	X	√	*Tubifex* reproduction showed strongest correlations with benthic metrics supporting cause-effect linkages. *Chironomus* growth showed some linkage with benthos but the direction of the correlation was inconsistent with impact.
Myra Falls	√	X	√	Benthic indicators (harpacticoids and *Pisidium*) were correlated with toxicity test results for *Tubifex* reproduction (positive correlation) and for *Hyalella* mortality (negative correlation). Chironomid mortality was not correlated with benthic indicators.

NT - Not tested.
√ - Effect demonstrated.
X - Effect not demonstrated.
? - Effect partially demonstrated.
* - More effective at demonstrating an effect.

TABLE 3.15: COMPARISON OF UNIT COSTS FOR SEDIMENT TOXICITY TESTING

Tool	Cost
Hyalella azteca	$600.00
Chironomus riparius	$600.00
Tubifex tubifex	$800.00

the multi-species tests generally all use benthic invertebrates and ones that fall within the same functional feeding group (i.e., gatherers); therefore, there is very little additional information gained from this battery of tests.

3.5.2 Fish Health and Community Indicators

Fish health (growth, organ size, fecundity) and community indicators (CPUE, BPUE) were evaluated at three of the four mine sites, with no fish sampling completed at Myra Falls because of a lack of fish in Myra Creek and negligible effects on water quality in Buttle Lake. Sentinel fish species at the three mine sites sampled included:

- Heath Steele - juvenile Atlantic salmon and blacknose dace.
- Dome - yellow perch and pearl dace.
- Mattabi - northern pike and white sucker.

3.5.2.1 Fish Population and Community Indicators

Catch-Per-Unit-Effort (CPUE) and Biomass-Per-Unit-Effort (BPUE)

Catch-per-unit-effort and biomass-per-unit-effort were assessed at Heath Steele and Mattabi. At Dome, habitat characteristics and fish capture methods varied among areas, thereby preventing any comparative assessment among areas.

Catch-per-unit-effort and biomass-per-unit-effort were effective in responding to mine exposure at Heath Steele, particularly when all fish species were considered as a whole (Table 3.16). These measurements were less effective or ineffective when considered at an individual species level. Both CPUE and BPUE measures showed correlations with aqueous metal concentrations and with predicted *in situ* chronic toxicity. These responses also parallel benthic community level responses. Accordingly, the responses may be interpreted as associated with either a toxicity response and/or benthic effects (i.e., reduced food base).

At Mattabi, however, CPUE and BPUE showed no statistically significant differences between reference and exposure areas, and were ineffective measures of fish response, either at the individual species level or at the whole community level. This lack of effects could be because of the low aqueous metal contamination in the exposure area. When the effluent was fully mixed in the receiving environment only zinc slightly exceeded Canadian Water Quality

TABLE 3.16 SUMMARY OF EFFECTIVENESS RANKINGS FOR CATCH/BIOMASS-PER-UNIT-EFFORT

Mine Site	CPUE[1]		BPUE[2]		Comments
	Individual Species	All Fish	Individual Species	All Fish	
Dome Mine	NT	NT	NT	NT	Qualitative analysis. Not effective but due to habitat differences and introduced species.
Heath Steele	?	√	?	√	Fish CPUE (all taxa combined) was reduced with degree of exposure and exposure area means were lower than reference means.
Mattabi Mine	X	X	X	X	No significant effect of mine exposure on fish abundance.
Myra Falls	NT	NT	NT	NT	

[1] Catch-per-unit-effort (numbers).
[2] Biomass-per-unit-effort.
NT - Not tested.
√ - Effect demonstrated.
X - Effect not demonstrated.
? - Effect partially demonstrated.
* - More effective at demonstrating an effect.

Guidelines for the protection of aquatic life so it is possible that these tools were effective in demonstrating that the mine was not impacting fish resources in this area.

Costs

The measurement of CPUE/BPUE adds little incremental cost to fish surveys, other than the cost of simple record keeping to account for numbers, biomass and effort. In the case where a subset of sentinel species is monitored more intensely for other parameters (e.g., age, organ size), CPUE and BPUE determinations require enumeration and weight determination of other species present. As noted previously, simple length-weight measurements are completed with little effort or cost.

Other Considerations

Measures of catch (numbers, biomass) per unit effort are accomplished readily and coincidentally with sampling of fish for any purpose, and can provide meaningful data as long as methods remain comparable at all reference and exposure stations. The data must always be assessed in light of natural habitat conditions that can influence CPUE, such as the presence of barriers or other habitat factors that can influence the species and numbers of fish present at any location.

Conclusions

At one of the two sites sampled effectively for CPUE and BPUE (Heath Steele), both tools responded effectively to exposure. Also, CPUE and BPUE measurements are fundamentally important indicators of environmental health. Although only limited data were collected in the 1997 field evaluations to assess the effectiveness of these tools, CPUE and BPUE were important tools at the Heath Steele site in demonstrating the effects of mine discharges on the fish community.

3.5.2.2 Fish Health Indicators (Growth, Liver Size, Reproductive Indicators)

Only fish growth (length and weight at age) was measured in Heath Steele fish. Reproductive indicators were not measured as the Atlantic salmon were immature and blacknose dace were not in an appropriate condition for gonad measurements (they had recently spawned). Also, for conservation reasons, Atlantic salmon could not be sacrificed in adequate numbers for liver size measurements.

Growth

Fish growth responses appeared largely ineffective at all three mine sites with six fish species tested, with either no effect on fish size or, in four cases, exposed fish being larger than reference fish (Table 3.17). Indeed, even though effects were observed in the benthic food base at Heath Steele and Dome, and are recognized through historic studies to also be impacted in Bell Creek at Mattabi, these benthic effects have not been expressed in terms of lower fish growth,

In both sexes of yellow perch and female pearl dace at Dome, the greater growth rates in exposed fish were coincident with reduced gonad weights at body weight, suggesting a difference in the proportional allocation of energy between somatic growth and reproduction in exposed fish. This could potentially reflect an exposure-related effect.

Liver Size

Liver size results could potentially show an increase, a decrease or no change in response to mine effluent exposure. An increase in size could potentially occur in response to increases in hepatic detoxification function, disruption in lipid and glycogen metabolism, or simply owing to greater energy storage. A decrease in liver size could be interpreted as an energy storage depletion response. Accordingly, it is difficult to conclude whether a difference in liver size in exposed fish occurs in direct response to metal exposure or to a difference in food availability. Any reference-exposure difference in liver size is viewed as indicating a potential direct or indirect response to metal exposure.

At Dome, liver weight was greater in exposed pearl dace, with liver weight declining from near-field to far-field and then reference areas. When liver weight was adjusted for body weight there were no reference-exposure area differences. Liver weight at age was also greater in exposed yellow perch than in reference perch. However, there were no liver size differences when adjusted for body weight instead of age.

At Mattabi, liver weight at age and at body weight were greater in exposed male white sucker compared to reference fish, whereas in pike, liver weight was lower in both age and size-

TABLE 3.17: SUMMARY OF EFFECTIVENESS RANKING FOR FISH HEALTH INDICATORS

Mine Site	Fish Health Indicators Fish Growth	Liver Weight	Gonad Weight	Fecundity	Comments
Dome Mine	X	?	?	?	Significant increase in length and weight of perch at age in exposure area. Significant increase in length and weight in exposure area for pearl dace. In yellow perch, no significant reference-exposure difference in gonad weight (males and females) and fecundity at age. Livers significantly larger in exposed yellow perch. Gonad weight (body weight adjusted) for males and females lower in exposure area. Liver weight adjusted for body weight showed no change. Significantly larger pearl dace gonad and liver weights in exposed females and males. Pearl dace fecundity higher in exposure area. Female dace body weight-adjusted gonad weight and fecundity lower in exposure area. Liver weight unchanged when adjusted for body weight.
Heath Steele	X	NT	NT	NT	Young-of-year (YOY) salmon were smaller in high density reaches (below barriers). Effect persists at later ages. No impairment of growth in salmon or blacknose dace.
Mattabi Mine	X	?	X	X	No significant differences in growth of white sucker. Significantly larger pike in exposure area. White sucker liver significantly larger in exposure fish. Gonad weight (body-weight adjusted) slightly smaller in exposed male sucker. Liver and gonad weight and fecundity in pike all significantly higher in exposed fish. Effects of exposure in pike not consistent with adverse impact.
Myra Falls	NT	NT	NT	NT	

NT - Not tested.
√ - Effect demonstrated.
X - Effect not demonstrated.
? - Effect partially demonstrated.
* - More effective at demonstrating an effect.

adjusted females in the exposure area. The tool is ranked as partially effective because these differences occurred (Table 3.17). However, because effects were seen in one sex of fish only and because the differences were opposite in direction in the two species, it is tenuous to interpret these differences as mine-related.

Gonad Weight and Fecundity

Gonad weights in Dome pearl dace and yellow perch (age-adjusted) were ineffective in responding to mine exposure, with greater gonad weights in exposed fish observed in pearl dace and comparable gonad weights in both exposed and reference yellow perch. Fecundities (numbers of eggs) showed the same patterns as gonad weights. Measures of gonad weight and fecundity are generally adjusted for body weight to adjust for any differences in size in order to place the variables on a relative scale. Therefore, when adjusted for body weight, which was greater in exposure areas, gonad weight was lowest in the near-field (female pearl dace) and lake exposure area (female and male yellow perch). Body weight-adjusted fecundities were also lowest in near-field pearl dace, but reference and exposure fish remained not significantly different for yellow perch. Thus, when considered in combination with body size, which is the more appropriate measure, effects in Dome pearl dace and yellow perch suggest that exposed fish allocate proportionally more energy towards somatic growth and less towards development of reproductive tissues than do reference fish. Based on this Dome result, gonad weight and fecundity are ranked as partially effective (Table 3.17).

At Mattabi, gonad weights and fecundities, after appropriate adjustment for body weight or age, showed no differences between reference and exposure areas in northern pike and white sucker.

Costs

Measurement of length, weight, age, liver weight, gonad weight and fecundity is inexpensive relative to tissue metal analysis. Length and weight determinations are made in the field and when fish are processed in batches of several specimens, they require only about one minute of time for two technicians (one to measure, one to record). The other measurements require about 10 minutes per fish for two people to dissect fish specimens to extract and label ageing structures and remove, weigh and prepare gonads and liver. Generally, a 10-minute time frame is required for fish such as white sucker that have a diffuse liver throughout their intestine, whereas fish with discrete livers such as pike require less processing time.

Typical laboratory costs for processing of ageing structures are approximately $10 per fish. Fecundity measurements can be typically completed for approximately $10 per fish. Accordingly, once captured and processed in batches of several fish, costs per fish for determination of fish health parameters is relatively low (in the order of $35-$45 per fish).

Other Considerations

Basic descriptive information on fish size and age is normally collected in fish monitoring studies, and is often required in the evaluation of metal bioaccumulation data (e.g., mercury concentration typically increases in fish muscle with increase in body size). Also, if organ size and reproductive health measurements are taken, age and/or body size adjustment of the data is often required prior to data analysis and interpretation.

The fish health indicators are not specifically compared to one another in terms of effectiveness in hypothesis testing, nor should they be compared because they measure different possible effects and should be considered as a whole. For example, greater somatic growth at the expense of reproductive tissue development produces a different conclusion (e.g., possible adverse effect) than does measurement of greater somatic growth in isolation (e.g., possible neutral or beneficial effect).

Small fish were effectively dissected for organ and gonad size determinations at Dome (pearl dace). However, as fish become smaller, the inherent error in weight determinations becomes greater owing to factors such as the sensitivity of the balance used and the moisture of the sample. From a practical standpoint, some fish smaller than about 100 mm fork length are less readily processed for organ measurements (e.g., species with diffuse livers).

Conclusion

Fish health indicators are of potential value in routine environmental effects monitoring at mine sites, based on evidence from the Dome Mine that fish may potentially allocate their available energy differently in response to exposure. As responses did not clearly occur at all sites (Mattabi) and other factors such as competition may be more important (Heath Steele), these tools must be used and interpreted with caution in monitoring programs. Regardless of their variable and uncertain effectiveness at the 1997 field sites, these basic parameters are fundamental indicators of effects on overall fish health, are relatively low in cost and are typically included in monitoring programs and baseline assessments at mines.

3.5.3 Benthic Community Health Indicators

Assessing the cost-effectiveness of benthic invertebrates in demonstrating mine-related effects was not a requirement of the 1997 field evaluations. However, because fish were not collected at some sites (e.g., Myra Falls) for testing of Hypothesis H6, benthic invertebrates were substituted. The objective of testing Hypothesis H6 using benthic invertebrates was not to evaluate the effectiveness of individual benthic measures (e.g., EPT index, % chironomids etc.) but was simply undertaken to show that when a selection of these measures is applied, benthic invertebrate assessments can be an effective monitoring tool for assessing mining-related impacts.

When considered individually, benthic community health indicators such as total density, number of taxa, EPT (Ephemeroptera-Plecoptera-Trichoptera) index at the generic level and abundances of site specific indicator taxa varied in their effectiveness from mine to mine (Table 3.18). However, on balance, benthic monitoring tools were usually effective or partially effective at the four sites, and the results indicate that several response indices should be determined to identify effects. Of those benthic parameters responding to exposure, responses tended to correlate well with either sediment metal concentrations (Dome, Mattabi, Myra Falls-Buttle Lake) or with aqueous metal concentrations (Heath Steele, Myra Falls-Myra Creek) (refer to Tables 3.3 and 3.6).

Other Considerations

The 1997 field program did not evaluate various benthic sampling or processing procedures that may influence cost-effectiveness. However, the samples were processed using two mesh size sieves (500 µm and 250 µm) to allow for more detailed evaluation outside of the scope of this project. Costs for processing 250 µm mesh samples are about 50 to 75% greater than the costs associated with processing of coarser mesh benthic samples. Previous studies at Heath Steele (BEAK 1997) and Mattabi (BEAK, 1993) indicate that benthic effects could also be discerned at these sites using a 500 µm mesh screen. Additional evaluation of the data sets will be undertaken to come to a definitive conclusion on the cost-effectiveness of different mesh sizes for demonstrating mine-related effects.

TABLE 3.18: SUMMARY OF EFFECTIVENESS RANKINGS FOR BENTHIC COMMUNITY HEALTH INDICATORS

Mine Site	Benthic Invertebrate Tools: Total Benthic Density	No.of Taxa	EPT Index	Abundance of Indicator Taxa	Comments
Dome Mine	√	√	√	√	Benthic indices such as number of taxa, density, EPT index (generic level) and indicator taxa all showed significant mine-related trends.
Heath Steele	X	?	√	√	Number of benthic taxa showed no linear trend in exposure zone. Number of EPT taxa was reduced with exposure. No spatial trends evident in total density. Rheocricotopus dominance showed a linear trend in exposure area and an exposure/reference mean difference. Orthoclad dominance showed a trend in the exposure area.
Mattabi Mine	√	√	NT	√	Significant decrease in exposure area, and exposure area trends in density, number of taxa and indicator taxa.
Myra Falls: • Stream • Lake	 X X	 X X	 √ NT	 √ √	Key indicator taxa abundances responded to exposure, including EPT index, Ephemerellidae, *Cricotopus* + *Orthocladius* and total chironomid abundance in creek, *Pisidium* and harpacticoid abundances in Buttle Lake.

NT - Not tested.
√ - Effect demonstrated.
X - Effect not demonstrated.
? - Effect partially demonstrated.
* - More effective at demonstrating an effect.

Testing of Hypothesis H6 showed that, when a selection of benthic measures is used, mine-related effects on the communities can clearly be established. However, as Table 3.18 shows, the use of only number of taxa and total density can be inadequate to demonstrate mine-related effects. This was even found to be the case at Heath Steele where impacts on the benthic community were quite obvious. The reason why these two measures by themselves are often inadequate to demonstrate impacts is because in environments where impacts are slight to moderate, sensitive taxa are replaced by more tolerant taxa and the area still maintains its carrying capacity (i.e., the total number of organisms that the food base can support). For example, this was quite evident at Heath Steele and Myra Creek where the EPT index at the generic level clearly showed a significant mine-related effect (i.e., less of the sensitive taxa in the exposure area) yet there was little or no change in the total number of taxa.

3.5.4 Effluent Toxicity

Chronic effluent toxicity was evaluated at three of the four 1997 mine sites - Heath Steele, Myra Falls and Dome Mine. Samples from Heath Steele were collected from the location in the river most impacted by ARD (acid rock drainage) rather than from the treated effluent, as the former location was more substantially affected by metals than was the latter. Myra Falls samples were collected from the treated effluent, although metal (zinc) loadings to the environment were dominated by uncollected seepage losses. The Dome Mine stopped discharging effluent two months before the field survey which was completed in October to better assess reproductive responses in fish. Therefore, any effluent toxicity effects in the receiving environment were not expressed at the time of the field sampling although residual biological response may be expected to persist after annual shutdown of the discharge. One of the samples collected at Dome (20 October 1997) was from the holding ponds and did not represent effluent that was discharged to the environment.

Hypothesis 13, which examines the relationships between effluent toxicity results and biological responses in the environment, was tested quantitatively only at Heath Steele. At Dome and Myra Falls, the linkages between toxicity test results and biological responses could not be effectively analyzed in the same quantitative manner as at Heath Steele. At Myra Falls, only one level of effluent exposure in the creek was available for biological sampling, precluding correlation analysis, and the dominant source of metals was not the effluent. At Dome, in addition to the fact that the mine was not discharging at the time of the survey, the percent effluent concentrations in the near-field and far-field fish collection sites were nearly the same. This produced a situation similar to the single exposure level correlation described above for Myra Falls. Accordingly, H13 was evaluated only qualitatively at the Dome and Myra Falls mines.

The four chronic effluent tests completed were:

- the *Selenastrum capricornutum* growth test;
- the *Ceriodaphnia dubia* survival and reproduction test;
- the fathead minnow survival and growth test; and
- the *Lemna minor* growth test.

In general, the test sensitivities at each mine ranked as follows:

Heath Steele: *Selenastrum* > *Ceriodaphnia* > *Lemna* ≈ fathead minnow
Dome Mine: *Ceriodaphnia* ≈ *Lemna* > *Selenastrum* ≈ fathead minnow
Myra Falls: *Ceriodaphnia* ≈ *Selenastrum* ≈ *Lemna* > fathead minnow

Thus, the *Ceriodaphnia* or *Selenastrum* tests were the most sensitive and the fathead minnow test the least sensitive to exposure to mine effluent at these three sites.

The effectiveness of the various effluent toxicity tests in producing results consistent with observed *in situ* effects in natural biota is presented in Table 3.19 for all sites tested. In general, all tests were effective in at least some cases. The fathead minnow test was ineffective at Myra Falls owing to insensitivity to the effluent (no chronic or acute effects). Specific observations of interest include:

- Toxicity occurred in fathead minnow at concentrations in the receiving environment where fish CPUE and BPUE were affected.
- Thresholds for growth impairment in fathead minnow in Dome effluent occurred at concentrations greater than those occurring downstream under effluent discharge conditions. No impairment of growth (body weight) occurred in yellow perch or pearl dace downstream of Dome.
- Benthic community effects occurred at Dome, Myra Falls and Heath Steele at effluent exposure concentrations consistent with sublethal and/or lethal effects in invertebrate (*Ceriodaphnia*) tests.

TABLE 3.19: EFFECTIVENESS OF CHRONIC TOXICITY TESTS IN CORRESPONDING WITH *IN SITU* BIOLOGICAL EFFECTS

Mine Site	Effluent Toxicity Tools				Comments
	Ceriodaphnia	*Selenastrum*	*Lemna*	*Pimephales*	
Dome Mine[1]	√	√	√	?	Effluent toxicity tests appeared effective in predicting effects on benthic communities and in predicting that there would be no effects on fish growth. Fathead minnow test was not effective in predicting body weight-adjusted effects in fish.
Heath Steele	√	√	√	√	Fish CPUE, BPUE and number of taxa decrease with predicted water toxicity to algae, *Ceriodaphnia*, duckweed or fathead minnow. Dominance of pollution-tolerant chironomids increases with predicted water toxicity. Other benthic indices not correlated with predicted toxicity. The four toxicity tests produce similar biology vs predicted toxicity correlations, when effluent is sublethally toxic. Fathead minnow test is less sensitive than other tests.
Mattabi Mine	NT	NT	NT	NT	
Myra Falls[1]	√	√	√	X	Benthic effects were observed in the exposure area of Myra Creek, and occurred at aqueous metal concentrations producing chronic toxicity in *Ceriodaphnia*, *Lemna* and *Selenastrum*. Therefore, these tests appeared to effectively predict benthic effects. No fathead minnow response occurred in any test (lethal or sublethal).

NT - Not tested.
√ - Effect demonstrated.
X - Effect not demonstrated.
? - Effect partially demonstrated.
* - More effective at demonstrating an effect.

[1] Conclusions based on qualitative testing of hypothesis; for Dome, testing is qualitative because biological sampling completed under zero effluent discharge condition.

The plant test results were consistent with effects observed in biological communities in the receiving environment.

Costs

Typical unit costs for three of the four sublethal tests were provided based on a recent survey of CAEAL (Canadian Association for Environmental Analytical Laboratories) and MEF-accredited (Ministère de l'environnement et de la faune, Gouvernement du Québec) toxicity testing laboratories in Canada for the Pulp and Paper EEM Program. Average test prices for small sample numbers (<5 samples per test) are:

Ceriodaphnia	$ 977
Selenastrum	$ 513
Fathead minnow	$1,072

The unit price for the *Lemna* test completed in this study was $500 (Saskatchewan Research Council).

Conclusion

Overall, all tests evaluated were effective in producing results consistent with observed effects in some elements of the downstream biological communities. It is difficult to selectively identify some tests as better than others for EEM monitoring, given that testing is best completed with a selection of species representative of various trophic levels in the environment. It may be argued that completion of two aquatic plant tests (*Lemna, Selenastrum*) would be somewhat redundant in an EEM program because, on balance, both were similar in sensitivity and effectiveness and have similar costs. The *Selenastrum* test as well as fathead minnow and *Ceriodaphnia* tests are widely commercially available, whereas the *Lemna* test method used in this study is relatively new and is only available at a few commercial laboratories. Environment Canada has recently prepared a standardized *Lemna* test method which is based on the methods applied in this study. Because the *Selenastrum* test is currently a requirement for the pulp and paper EEM program, coupled with the fact that it has relatively the same sensitivity as the *Lemna* test in assessing mine effluents, it would be the preferable selection for routine EEM programs.

3.6 Are Contaminants Causing the Responses?

As indicated previously, this question is not answered directly through the application of specific monitoring tools evaluated in this study, or through any of the hypotheses tested. Rather, the question is evaluated only by a weight-of-evidence provided by affirmative responses to the first three questions, and particularly by the strength of correlations between exposure indicators (chemical concentrations) and biological responses in hypotheses H9 through H13.

At all four mine sites, evidence clearly indicated that contaminants were getting into the system (Tables 3.2 and 3.5) and were bioavailable based on bioaccumulated metals in fish and effluent and sediment toxicity data (Tables 3.8, 3.12 and 3.19), and that certain biological responses were correlated with metal concentrations in the environment (Tables 3.3 and 3.6).

It was obvious that bioavailability of metals, especially in sediments, varied among sites. For example, at Mattabi sediments were much more contaminated with metals compared to sediments at Myra Falls and at both sites water quality parameters were below CWQG. If sediment toxicity can be considered to reflect bioavailability of metals in sediment, sediment metals at Mattabi are low in bioavailability (i.e., sediment toxicity was not strongly related to metal concentrations). This absence of strong metal-related toxicity is unusual, given the very high concentrations of metals such as zinc in the sediments. There was very little acute sediment toxicity at Mattabi and severe acute toxicity at Myra Falls.

A number of benthic community and fish population responses and bioaccumulated metals in tissues were correlated with sediment and water concentrations of metals. The directions of exposure-response relationships were often consistent with biological effects due to mine-related contaminants. Furthermore, *in situ* toxicity predicted from laboratory toxicity testing also reflected biological effects in most cases. Accordingly, the field data support a conclusion that contaminants appear to be causing the responses. Mattabi was the only site where toxicity responses to contaminants were weak, despite high total metal levels.

It is important to consider that dose-response relationships in the field do not necessarily prove cause and effect. Rather, a combination of controlled laboratory testing of metal toxicity and field evidence such as provided by the 1997 field evaluations would be appropriate to provide further detail on cause and effect (e.g., which metals individually or in combination produce a response at each site).

Sediment Quality Triad

The sediment quality triad also uses a weight-of-evidence approach to suggest if contaminants are causing the response. The triad was evaluated at three sites. At all three sites, some significant sediment chemistry-benthic community, chemistry-toxicity and toxicity-benthic community correlations existed (although not all were mine-related). Moreover, there was a dominant mine-related gradient in sediment quality at all three sites, with lower moisture and organic content and higher metal concentrations in the near-field.

Dome Mine

The analysis of the sediment quality triad at Dome Mine showed that, using the dominant sediment quality and benthic community gradients, the linkages were strong between sediment chemistry and toxicity and between toxicity and the benthic community response. However, the linkage between the dominant sediment quality gradient and the dominant benthic community gradient was not strong; the benthic community was mainly driven by a secondary gradient in sediment quality (one not clearly related to Dome Mine). Results also suggested that the causes of benthic and toxicity responses may be different; habitat effects are an important influence on the benthic community. Overall, the analysis shows that, as a group, sediment toxicity and benthic community tools were responsive to sediment quality conditions.

Mattabi Mine

The sediment quality triad produced either weak or insignificant linkages between sediment chemistry, sediment toxicity and sediment biota, depending on the statistical solution and data sets used. The strongest linkage was found using partial metal concentrations, which supports the intended use of partial metals in that they may better reflect bioavailable sediment contaminants. Using the dominant sediment quality (partial metals only) and benthic community gradients, there was a strong linkage between sediment quality and benthic community composition; however, the linkages between sediment chemistry and toxicity, and benthos and toxicity, were not strong. Sediment toxicity (i.e., *Tubifex* reproduction) was related to growth-promoting features of Mine Creek Bay sediments, and was correlated with Mine Creek Bay benthic species. One may conclude that sediment metals are relatively low in bioavailability, to such an extent that other factors (e.g., minor grain size differences, nutritional features, etc.) tend to mask any impacts of sediment-associated metals.

Based on the above considerations, the Mattabi example serves to illustrate that metals released to the environment from mining activities can in some cases produce little measurable toxicity, despite the presence of high concentrations in sediments and water. The results from this site demonstrate why it is critical to primarily focus on biological responses in an EEM program rather than relying primarily on sediment and water chemistry measurements.

Myra Falls

The analysis of the sediment quality triad at Myra Falls showed that overall, linkages were strong between sediment chemistry and both benthic community and sediment toxicity responses. Using the dominant sediment quality and benthic community gradients, there was a strong linkage between sediment quality and benthic community composition, and between sediment quality and toxicity; however, the linkage between benthos and toxicity was not so strong. A secondary benthic community gradient was better related to the toxic response at certain near-field stations. Overall, the analysis shows that, as a group, sediment toxicity and benthic community tools were responsive to sediment quality conditions.

Based on the outcome of the sediment quality triad analysis at the three sites, it appeared to be an effective means of identifying whether mine-related contaminants or other sediment features were linked to the biological responses.

4.0 SUMMARY

Table 4.1 compares the effectiveness of alternate tools that may be used to measure metal concentrations, metal bioavailability or biological responses and identifies the preferred tool based on the outcome of hypotheses testing for the 1997 Field Evaluations.

Some of the tools evaluated were effective at demonstrating an effect, whereas others were not. It is important to note that tools that did not demonstrate a mine-related effect may not necessarily be ineffective. The limited effectiveness of some of these tools may be due to low metal bioavailability (e.g., Mattabi Mine) or other confounding factors (e.g., no effluent discharge during sample collection, habitat variability, other sources of contaminants).

Of the tools in the same tool box ranked as effective (e.g., dissolved and total metals, total and partial metals), major differences in effectiveness were not evident at the mine sites. Therefore, the costs of each tool and other considerations involving the tool were important in the selection of which was considered to be the most cost-effective monitoring technology.

The tools that were considered to be the most cost-effective are as follows:

Tool Box	Tool
• Water Chemistry:	total metals (tentative); conditions at the four mines not conducive for rigorous evaluation of the relative effectiveness of total and dissolved metals
• Sediment Chemistry:	total metals
• Sediment Toxicity:	*Hyalella azteca*
• Benthic Invertebrate Assessments:	effective at all sites and appeared more sensitive at demonstrating mine effects than fish health indicators
• Fish Tissue Metals:	muscle and liver
• Fish Tissue Metallothionein:	no conclusion regarding most effective tissue type(s)
• Fish Tissue Metals versus Fish Tissue Metallothionein:	fish tissue metals
• Fish Health Indicators:	data variable, but concluded that fish measurements were helpful in demonstrating mine-related effects or a lack of effects
• Effluent Toxicity:	*Ceriodaphnia dubia* (water flea), *Selenastrum capricornutum* (algae), *Pimephales promelas* (fathead minnow)
• Periphyton:	only evaluated at Heath Steele, data too limited to base a conclusion on its effectiveness as a surrogate for sediments in erosional habitats

TABLE 4.1: COMPARATIVE EFFECTIVENESS OF MONITORING TOOLS

Tools	Comparison	Preferred Tool
Total Metals vs Dissolved Metals in Water	**Dome:** Total and dissolved metal concentrations approximately equal in reflecting elevated metal concentrations. Concentrations of both appeared unrelated to biological effects, although some correlations occurred between metal concentrations and tissue response. **Heath Steele:** Dissolved metal and total metal concentrations were similar in terms of strength of correlation with biological responses. **Mattabi:** Total and dissolved metal concentrations approximately equal in reflecting elevated metal concentrations. Concentrations of both appeared unrelated to most biological effects, although some correlations occurred between metal concentrations and tissue response. **Myra Falls:** Dissolved metal concentrations were similar in effectiveness to total metals in predicting benthic responses in Myra Creek. Assessed qualitatively.	Total metals (tentative): • lower cost • lower probability of contamination • can compare to existing guidelines • further testing of dissolved metals at a site having greater TSS levels or more variable dissolved metal fractions would be useful
Total Metals, Partial Metals and SEM/AVS in Sediment	**Dome:** Total and partial metals were, on average, comparable in reflecting benthic effects and toxicity effects. The SEM/AVS ratio was unrelated to benthic effects or sediment toxicity at this site. **Mattabi:** Total and partial metals were, on average, similar in reflecting benthic effects, with total metals slightly better correlated than partial metals. Total metals also better reflected mine gradients (showing “contaminants getting into the system”). The SEM/AVS ratio was unrelated to benthic effects or sediment toxicity. **Myra Falls:** Total metals were, on average, slightly better correlated with benthic effects and sediment toxicity than were partial metals. The SEM/AVS ratio was not correlated with benthic effects or sediment toxicity.	Total metals: • more effective • lower cost • can compare to existing guidelines
Sediment Toxicity Tests	**Dome:** *Hyalella* test was effective in reflecting mine-related impact. **Mattabi:** None of the tests indicated mine-related impact (i.e., reference-exposure differences or exposure area gradients). Absence of mine-related sediment toxicity precluded more rigorous evaluation of tests. **Myra Falls:** *Hyalella* and *Chironomus* test results were more sensitive and better linked with sediment metals and benthic effects than were *Tubifex* test results. However, some reproductive responses in *Tubifex* were effective in showing exposure effects.	*Hyalella azteca:* • most sensitive • better correlations with metals and benthos • lower cost than *Tubifex* and similar to *Chironomus*
Benthic Community Health Indicators (density, no. of taxa, indicator taxa)	**Dome:** Several indices were effective in reflecting mine-related impact including total density, no. of taxa, EPT and abundance of indicator taxa. **Heath Steele:** Benthic EPT index and abundances of some indicator species were more responsive than numbers of taxa to exposure. Total densities did not respond effectively. **Mattabi:** Several indices indicated mine-related impact including total density, no. of taxa, and abundance of Hydracarina, *Chironomus* and *Pisidium*. Of these indicators, number of taxa and % Hydracarina were most strongly correlated with sediment metal concentrations. **Myra Falls:** Abundances of indicator taxa responded effectively to mine effects in both lake and stream environments, although different indicators were effective in lake than in stream habitat. Other indices (total densities, numbers of taxa) were marginally effective or ineffective. The EPT index was effective in Myra Creek.	Effective at all sites. Should use a suite of benthic measures.

Tools	Comparison	Preferred Tool
Fish Tissues - Metals	**Dome:** Yellow perch muscle was superior in indicating mine exposure compared to other tissues used for perch. Pearl dace viscera was most effective in showing mine-related trends moreso than perch tissues. **Mattabi:** In sucker and pike, no tissue was clearly superior in responding to mine exposure overall. In sucker, gill was the only tissue responding to any of the major Mattabi metals. In pike, gill, kidney and muscle all responded in terms of some of the major metals. Liver responded effectively to bioaccumulated Cu+Cd+Zn and Cu in pike, but metal source in liver unrelated to mine exposure.	Muscle and Liver: • muscle provides link to human health • liver used historically to monitor water quality
Fish Tissues - Metallothionein	**Dome:** MT did not show a mine-related response in any tissues. **Mattabi:** In sucker, MT levels did not respond to mine exposure in any tissues. In pike, gill and kidney were responsive to exposure. Pike liver from all reference-exposure fish responded to accumulated metals (Cu, Cd+Cu+Zn) but response unrelated to mine exposure.	No conclusion
Fish Tissues - Metals vs Metallothionein	**Dome:** MT did not respond to exposure. Metals in perch muscle and pearl dace viscera were more effective. **Heath Steele:** On balance, MT responded more frequently or strongly to exposure than did metals in small fish viscera. Effectiveness differed greatly from species to species and MT only responded in caged salmon. The responsive metals variously included Zn, Pb, Cu and Cd. Gill metals were variable in responsiveness to exposure (Cd most effective); gill MT was effective. Gill concentrations of some metals were responsive to exposure. Visceral metals were not. Visceral and gill MT appeared equally responsive to exposure. **Mattabi:** In the only tissues where MT and metals both responded to exposure (gill and kidney in pike), MT and tissue metals responded similarly. For MT and tissue metals overall, effects were more often demonstrated for metals than for MT.	Fish Tissue metals: • more effective • lower cost
Fish Health Indicators	**Dome:** Among the responses examined (length, weight, liver weight, gonad weight, fecundity), only liver weight showed responses that could potentially represent effects, i.e., greater liver weight in exposed fish. However, when the reproductive measures were adjusted for body weight, mine-related effects were reflected in yellow perch (male and female) and female pearl dace. **Mattabi:** Among the responses observed (growth, condition, liver weight, gonad weight, fecundity), most were inconsistent with adverse mine effects. A small reduction in gonad size in exposed male sucker could be construed as a mine effect. Liver weight increases in both male sucker and in pike, gonad weight increases in pike and increased somatic growth in pike are all less obviously related to mine exposure.	Fish health measures were variable. Lack of effects may be due to low metal exposure and low metal bioavailability.
Effluent Toxicity	**Dome:** Effluent toxicity results were effective in predicting effects (*Ceriodaphnia* and *Selenastrum*) or lack of effects (fathead minnow) on benthic and fish communities. **Heath Steele:** All tests were effective in predicting in-stream effects on natural communities; fathead minnow test was the least sensitive of the four. **Myra Falls:** *Selenastrum, Lemna* and *Ceriodaphnia* tests were generally more sensitive than the fathead minnow test. Fathead minnow test was ineffective.	*Selenastrum, Ceriodaphnia,* fathead
Metals in Water vs Metals in Periphyton	**Heath Steele**: Metals in water rather than periphyton were more strongly correlated with community level biological responses on balance. Periphyton Cu, Zn and Pb were better correlated with visceral metals in blacknose dace than were aqueous metals, possibly reflecting periphyton in the gut.	Insufficient data for a conclusion.
Fish CPUE/BPUE (individual species vs whole community)	**Heath Steele:** CPUE and BPUE were responsive to exposure; CPUE and BPUE were more responsive at the community level (all fish) than at the individual species level. **Mattabi:** Neither CPUE, BPUE nor numbers of fish taxa were responsive to mine exposure.	Insufficient data for a conclusion.

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