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MERCURY RELEASE IN

HYDROELECTRIC RESERVOIRS

research report

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MERCURY RELEASE IN HYDROELECTRIC RESERVOIRS

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ABSTRACT

Mercury has been found to accumulate in fish to unacceptably high levels subsequent to the impoundment of a number of recently constructed hydroelectric reservoirs. Recognizing this potential problem, the CEA contracted ESSA Ltd. to carry out a scientific investigation of mercury release in hydroelectric reservoirs, with an aim towards developing methods of anticipating future problems and mitigating against existing ones. The study comprised several activities, including an extensive review of existing knowledge, workshops to plan the study and evaluate its results, laboratory microcosm experiments, field data collection, and the collection and analysis of existing data relevant to the mercury problem. The results of the study suggest that numerous factors play a role in determining the degree of elevation of mercury levels in biota following impoundment, and that the relative importance of different factors varies among regions. This makes it virtually impossible to develop a general model for predicting the expected magnitude of a mercury problem at a proposed site. A set of simple guidelines are provided, however, that can be used to provide some idea of the risk of a problem arising at a proposed site. As well, options for mitigation are very limited, with regulations of fish harvests offering the only reasonable course of action in the majority of instances.

Keywords: Mercury bioaccumulation; hydroelectric reservoirs

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CAVEAT

The choices of statistical analysis techniques, and the conclusions regarding the results of the data analyses presented in this report, are solely the responsibility of the authors, and do not necessarily reflect the views of the people or organizations that supplied the data. Some of the data we have used were released to us under the condition that it be used only for the work described herein, for the Canadian Electrical Association.

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1 INTRODUCTION

1.1 BACKGROUND

Mercury pollution in the aquatic environment has received a great deal of attention over the last decade, particularly since the incidence of mercury poisoning at Minamata Bay, Japan. Within Canada, effluents from a chlor-alkali plant and pulp mill resulted in mercury contamination of the English-Wabigoon River system, threatening commercial fish operations and traditional food supplies of native groups (Jackson et al., 1980).

Recently, another mercury problem has come to light. Newly impounded reservoirs (without any local anthropogenic sources of mercury) have been found to contain fish with high levels of this element (Knight and Herring, 1972; Potter et al., 1975; Abernathy and Cumbie, 1977; Meister et al., 1979; Bodaly and Hecky, 1979; Environment Canada, 1979; Waite et al., 1980). This phenomenon has been reported in reservoirs from various parts of Canada, including Labrador (Bruce and Spencer, 1979), northern Manitoba (Bodaly et al., 1984b), and southern Saskatchewan (Waite et al., 1980).

In recognition of this potential environmental problem, the Canadian Electrical Association decided in 1983 to sponsor a study of mercury release in hydroelectric reservoirs with the following objectives:

1) to identify sources and conditions most likely to result in mercury problems;

2) to summarize the state-of-the-art knowledge of mercury uptake processes;

3) to identify factors leading to greatest mercury bioaccumulation;

4) to develop methods for predicting possible mercury problems;

5) to provide guidelines for possible mitigative action; and

6) to recommend further research needs.

This report presents the results of this study, with emphasis on how the various activities of the study have served to address the objectives listed above. The report is divided into a number of sections, each devoted to a discussion of the methods and results of the major study activities. To introduce the reader to the remainder of the report, we first present a general overview of our approach to the study.

1.2 OVERVIEW OF THE STUDY APPROACH

The general approach to this study is based on a methodology known as Adaptive Environmental Assessment and Management (AEAM) (Holling, 1978). AEAM is a collection of tools and techniques which centre around the use of systems analysis and interdisciplinary workshops to conduct an analysis of an environmental issue which is both integrated and relevant. In this application, we chose to use the development and evaluation of a model of mercury in newly impounded reservoirs as the guiding force for the entire project. A conceptual model was developed at an interdisciplinary workshop held near the start of the project (Section 3), and then refined, quantified and evaluated on the basis of research conducted subsequent to the workshop. A second workshop provided a forum for critique of the resulting model, and for a focussed discussion of other project results. In particular, this second workshop offered an opportunity for debate amongst specialists on the implications of the study regarding objectives (5) and (6) above, namely options for mitigation and future research needs.

The research activities conducted subsequent to the first workshop fall into three major categories:

1) laboratory microcosm experiments to look at specific cause-effect relationships (Section 4);

2) field data collection from existing reservoirs to test broader, more empirical hypotheses, and to supplement existing data (Section 5); and

3) collection and analysis of existing data on mercury in reservoirs (Section 6).

Data collected from the field were combined with existing data to form a single database for statistical analysis (Section 7).

Finally, an extensive review of the literature relevant to the subject of mercury uptake by fish, particularly in recently flooded reservoirs, was conducted at the outset of the study and then revised as necessary near the study's completion. This review helped to focus debate at the first workshop and ensured that any research conducted during the study would not amount to an unnecessary repetition of earlier work.

All of the activities of this study were, therefore, closely linked to each other. Figure 1 depicts the relationship between these activities and provides the reader with a useful frame of reference for the remainder of this report.

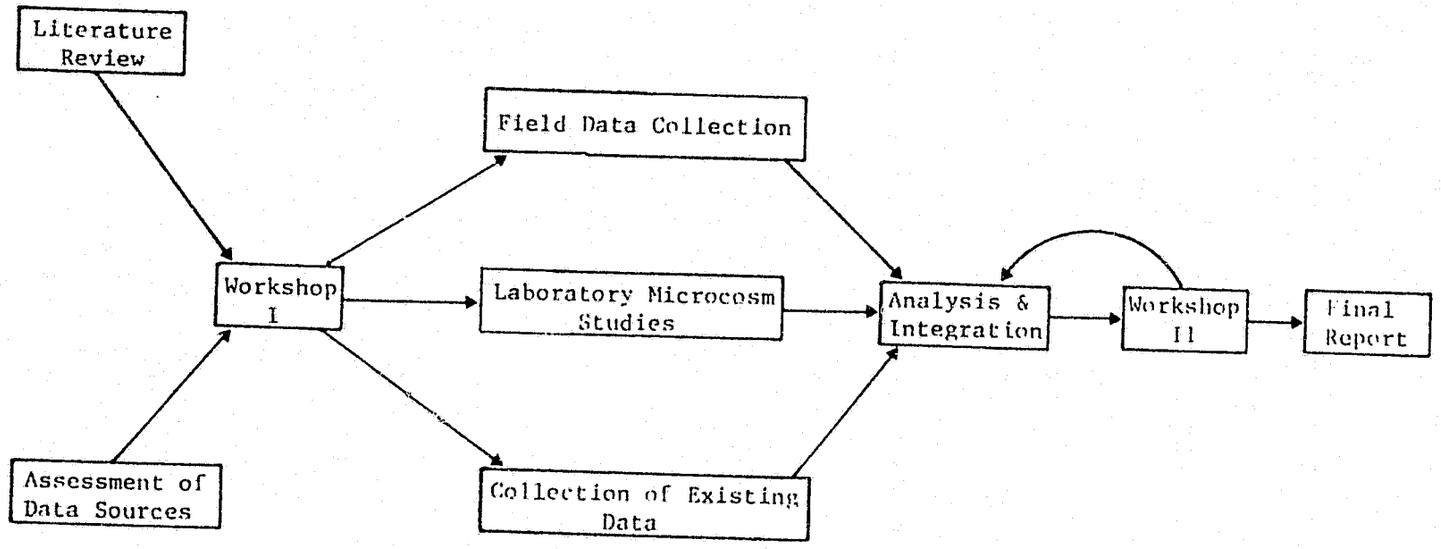


Figure 1 Overview of the major project activities and their relationship to each other.

2 REVIEW OF EXISTING KNOWLEDGE

2.1 INTRODUCTION

As discussed in Section 1, increased concentrations of mercury ("above background") have been identified in fish tissue from several newly-impounded reservoirs. The water bodies of concern are not considered to be polluted, being remote from known point sources of mercury, and with water column concentrations of total mercury less than 0.020 ug/L, which is close to the detection limit for routine standard analytical methods (Environment Canada, 1979). In this section, information from the published literature has been reviewed and organized into sections dealing with the following subject areas:

1) the biochemical behaviour of mercury;

2) the biogeochemical cycling of mercury;

3) the toxicology of mercury as it relates to human health;

4) the processes by which fish accumulate mercury;

5) published reports of elevated mercury in reservoir fish; and

6) possible mechanisms by which mercury is made available for bioaccumulation.

Two terms which are used throughout this report warrant definition before presenting our review of existing knowledge. Bioaccumulation is defined as the increased concentration of an element (or compound) in an organism over the concentration in the abiotic medium in which the organism lives. If the ratio of concentration in organism : concentration in medium is greater than one, bioaccumulation has occurred. In our text, bioconcentration is sometimes used synonymously with bioaccumulation. Biomagnification is defined as the increase in concentration of an element (or compound) in organisms with increasing trophic level in the food chain.

2.2 BIOCHEMICAL BEHAVIOUR OF MERCURY

Mercury is a member of group IIB of the periodic table, but it displays unique chemical properties which strongly influence its behaviour in biological systems. In particular, the high affinity of mercury for thiol (SH) groups and its corresponding low affinity for oxygen ligands means that mercury binds strongly to sensitive sites on proteins, especially enzyme proteins; this in large measure explains its extreme toxicity. The stability of mercury-carbon bonds in air or water, a consequence of the low affinity of mercury for oxygen, means that mercury compounds display strong covalent properties (i.e. they are stable and do not ionize), resulting in persistence of organo-mercurials in the environment.

For physico-chemical properties and potential biological effects, mercury can be conveniently considered in its three major forms: elemental (Hg^0), inorganic (Hg^+, Hg^{2+}), and organic (e.g. RHg^+, alkyl mercury). Elemental mercury is volatile, sparingly soluble compared with most compounds (but with high solubility compared with other metals in its group), and non polar. As such it can cross biological membranes readily. It direct effects on biota are documented only for occupational and experimental exposure (i.e. it is not of direct significance as an environmental contaminant). It can, however, be oxidized to Hg^+ or Hg^{2+} in the mercury cycle and its indirect effects are therefore not trivial.

Inorganic mercury salts show class B characteristics; salts such as $HgCl_2$ are strongly covalent, and predominantly undissociated in aqueous solution. An important feature of Hg^{2+} is its affinity for S and SH groups. However, inorganic mercury per se has low toxicity and a short biological half life. The significance of inorganic mercury in the environment is in the context of the mercury cycle, rather than for its direct effects on biota.

The toxicity of organic mercury compounds (organo-mercurials), including methyl mercury, reflects their physico-chemical properties. The stability of the Hg-C bond in aqueous solution, combined with the facility with which alkylation or inorganic mercury can occur in the environment, means

that alkyl mercury compounds are potent toxins of high risk. In the past, (and on a continuing basis in some parts of the world) organomercurials were produced and used as fungicides and slimicides in agriculture and industry. Once in biological systems, simple alkyl mercury compounds bind strongly, especially to S or SH sites, from which they are removed by chelates only with great difficulty. This binding largely accounts for the long biological half-life of methyl mercury in man (70 d compared with 6 d for inorganic mercury).

2.3 BIOGEOCHEMICAL CYCLING OF MERCURY

The discovery in the late 1960s that the process of mercury methylation could occur under a wide variety of environmental conditions (Jernelov and Lamm, 1973) marked a turning point in the study of the toxicology of mercury. The interconversions which are known to occur, shown in Figure 2, mean that no forms of mercury in the environment can be considered safe. Perhaps more than for any other metal, knowledge of conditions and factors which control the interconversions of the mercury cycle are crucial to assessing the risk of mercury toxicity to man and biota.

Methylation of mercury has been shown to occur in sediment (Jensen and Jernelov, 1969; Bisogni and Lawrence, 1975; Wood, 1979; Furutani and Rudd, 1980) and in the water column (Furutani and Rudd, 1980; Topping and Davis, 1981). Details of the proposed biochemical mechanism(s) of methylation are discussed by Craig (1980). The most significant biological methylating agent is methyl cobalamin (CH_3CoB_{12}), from which the methyl group is transferred to Hg:

$$CH_3CoB_{12} + Hg^{2+} + H_2O \rightarrow H_2OCoB_{12}^{+} + CH_3Hg^{+} \qquad [2.1]$$

A second methyl group can be added, and there is in fact uncertainty as to the nature of the major product, whether CH_3Hg^{+} or $(CH_3)_2Hg$. A free radical mechanism is an alternative to this transfer of a methyl group and has been proposed as a possible pathway for non-biological methylation:

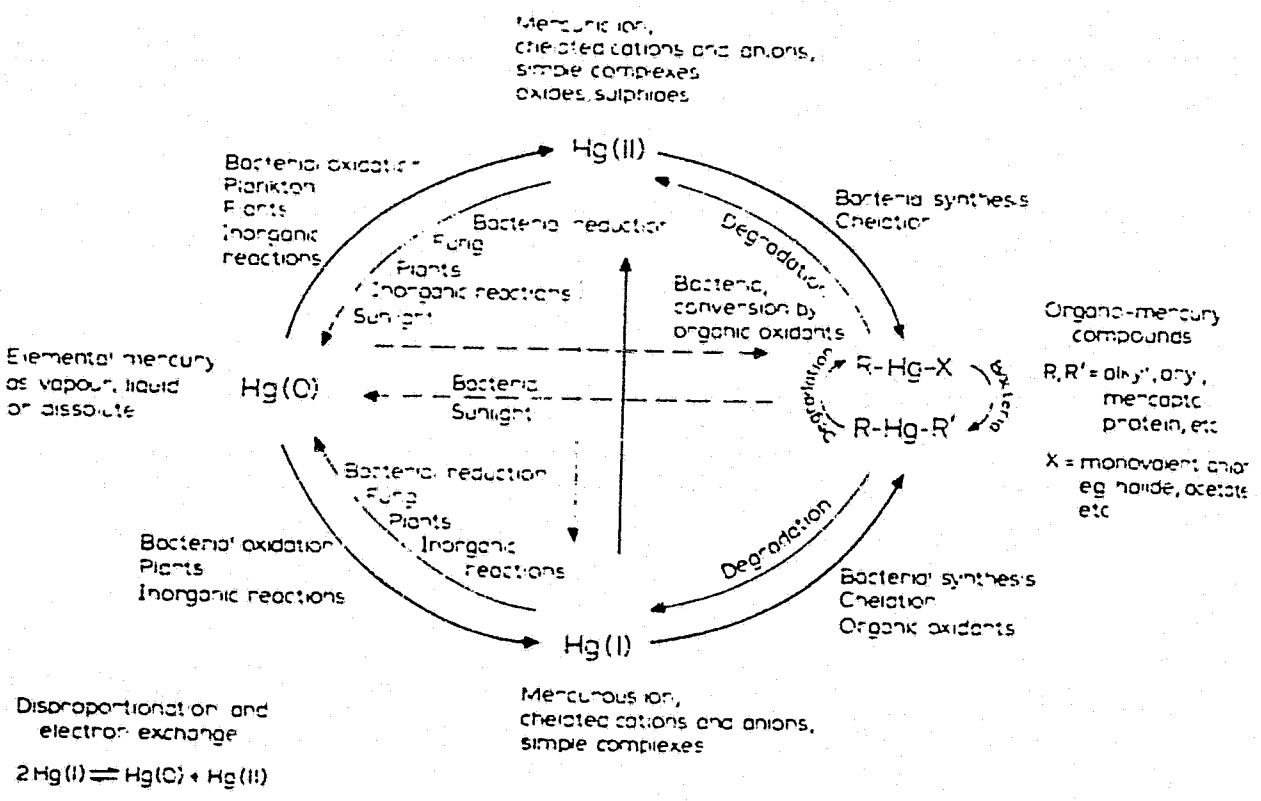


Figure 2 Biological/environmental interconversion of Hg(0), Hg(I), and Hg(II); from Carty and Malone (1979).

$$CH_3Co^{III}B_{12} + M^{II} \rightarrow CH_3M^{III} + Co^{II}B_{12} \qquad [2.2]$$

Microbial methylation can occur under aerobic or anaerobic conditions (Bisogni and Lawrence, 1975; Olsson et al., 1979). Factors which have been reported to affect rates of methyl mercury production include sediment type (Miller and Agaki, 1979), water temperature (Bisogni and Lawrence, 1975; Wright and Hamilton, 1982; Nagase et al., 1982), pH (Fagerstrom and Jernelov, 1972; Langley, 1973; Wood, 1979; Nagase et al., 1982), redox potential (Ridley et al., 1977; Wood, 1979), inorganic mercury concentration in sediments (Wright and Hamilton, 1982; Nagase et al., 1982), nutrient availability (Jensen and Jernelov, 1969; Fagerstrom and Jernelov, 1972; Wright and Hamilton, 1982; Nagase et al., 1982), bioturbation of the sediments by macroinvertebrates (Langley, 1973), and availability of a chemical precursor or methyl donor (Ridley et al., 1977; Wood, 1979). Nagase et al., (1982) reported that in aquatic environments, humic substances can provide a methyl donor and may stimulate methyl mercury production from the sediments; however, this conclusion was based on results obtained under very high mercury conditions (1 ug/L).

Sediment type, which includes proportion of organic matter and nutrients, appears to be a key factor in the rate of methyl mercury production from bottom sediments. The overall concentration of nutrients in mercury methylating sediments may not be as important as the amount of nutrients available for microbial activity near the sediment-water interface. This may explain why the covering of methyl mercury producing sediments with clean, nutrient-poor, sediments effectively reduces the production of methyl mercury (Jernelov, 1970; Rudd et al., 1983).

The amount of methyl mercury in water is actually the result of two concurrent processes, methylation and demethylation, both of which can be bacterially mediated (Brosset, 1981). While most field measurements of mercury methylation provide net methylation rates, and do not distinguish between the production and degradation of methyl mercury, recent attempts

have successfully demonstrated the respective response of the two types of process to certain environmental factors. Ramlal et al. (1985a) examined the effect of experimentally decreasing pH on methylation and demethylation in lake sediment. Methylation was shown to be more sensitive than demethylation to decreased pH, and ceased entirely in sediments below pH 5.0. In contrast, in the water column (epilimnion), methylation was enhanced by decreasing pH, while demethylation in the water column showed less sensitivity, but in fact was decreased by lowering or raising the pH (Xun et al., 1985).

It is widely accepted that mercury levels in biota are lower in more productive, or more eutrophic waters. The larger biomass per unit volume of water in more eutrophic lakes is thought to dilute the total amount of biologically available mercury which results in lower mercury levels in the biota (D'Itri et al., 1971; Fagerstrom and Jernelov, 1972; Beijer and Jernelov, 1979; Huckabee et al., 1979; Wood, 1979; Hakanson, 1980). Widespread support of the concept of biomass mercury dilution in more productive waters appears to have been perpetuated from an early report by D'Itri et al. (1971), with little factual or experimental evidence to support the notion. In the early study, D'Itri et al. (1971) compared mercury accumulation in only two ecosystems, and concluded that adsorption onto particulate organic matter in the more eutrophic lake, not biomass dilution, was the key factor in limiting mercury bioavailability to fish. In an experimental study, Rudd and Turner (1983) found that stimulating productivity by adding nutrients increased fish growth rates but did not lead to lower mercury levels in fish.

2.4 TOXICOLOGICAL AND HUMAN HEALTH CONSIDERATIONS

It has already been stated that there is a gap between our understanding of the biological chemistry of mercury, which is relatively good, and our understanding of the biological effects of mercury toxicity which is much poorer. In the present context, the effects of mercury will be considered only with respect to dose-response concentrations, and the rationale for existing standards for consumption of mercury contaminated

food. A number of reviews deal in detail with health effects of mercury (e.g. Miller and Miller, 1979; Piotrowski and Inskipp, 1981; Suzuki, 1979; Jugo, 1979; Berlin, 1983).

Organic mercurials are more toxic on a weight-for-weight basis than are the inorganic salts of mercury. Organic mercurials are characterized by rapid absorption, slow elimination and permanent effects, in contrast to inorganic mercury salts which are poorly absorbed, rapidly eliminated and effects which are not generally permanent. Jugo (1979) stated that "Of all alkylmercurials, methyl mercury represents the greatest challenge. Public concern is essentially reduced to the problem of methyl mercury".

For all animals, the principal manifestations of methyl mercury toxicity are abnormalities in the central nervous system. The syndrome in affected humans and animals is generally known as Minamata disease, named as a result of severe mercury poisoning which occurred in Minamata, Japan in 1956 (D'Itri and D'Itri, 1977). The fetus is at particular risk for Minamata disease since methyl mercury crosses the placental barrier and accumulates at 2-3 times the levels in the mother.

The route to man is through food and water, with food the most likely source of toxic doses of mercury. Most food contains < 0.002 mg/kg mercury (Rose, 1979) but fish tend to accumulate much higher levels. This results from the fact that mercury is biomagnified in food chains, especially aquatic ones, and that it bioaccumulates in aquatic organisms. Unlike other metals which tend to be concentrated in organs, bone or nervous tissue, methyl mercury also accumulates in the muscle of fish (i.e. the edible part), even though the brain is the organ where toxicological effects are manifested. Other food items which may contain high mercury levels include waterfowl, seal, walrus and whale. Studies in Canada have linked high mercury in blood and the brain to the diet of people hunting and fishing in regions with unacceptably high mercury levels in food items obtained from the aquatic food chain (Desai-Greenway and Price, 1976). Table 1 shows estimates of human exposure to methyl mercury from fish consumption. Populations at risk are clearly those who consume

Table 1

ESTIMATE OF EFFECTIVE HUMAN EXPOSURE
TO METHYL MERCURY FROM FISH CONSUMPTION

	Intake of methyl mercury from fish (ug/d) with different fish mercury concentrations				
Human consumption of fish (g/d) \ Concentration of mercury in fish (mg/kg)	0.2	0.5	1.0	2.0	5.0
Average	20	4	10	20	40	100
Elevated	100	20	50	100	200	500
High	300	60	150	300	600	1,500
Very high	1,000	200	500	1,000	2,000	5,000

The calculation given in Table 1 is aimed at estimating human exposure from fish, assuming levels encountered in practice and food consumption thought typical for selected groups of populations.

Note:

a) The threshold body burden (see text) would be reached with a daily consumption of about 200-500 ug methyl mercury per day.

b) The allowable intake, according to the Joint FAO/WHO Expert Committee on Food Additives, 1972 (200 ug per week), is equivalent to a daily intake of about 30 ug.

Modified from Piotrowski & Inskipp, 1981.

large amounts of fish. The average consumption varies widely from country to country and within countries depending upon income and dietary habits (Piotrowski and Inskipp, 1981).

In Sweden, Denmark, and Spain, average daily intake of fish is of the order of 40 g per day and in Japan it is as high as 100 g per day. Selected populations in the U.K., the U.S.A. and Sardinia have been estimated to consume 200-300 g per day and consumption for some native Canadian Indians was reported at 450 g per day (Barbeau et al., 1976). The highest consumers may take more than 1,000 g per day. Again, Canadian Indians working as tourist guides or fishermen had an estimated upper limit of 1,300 g per day (Barbeau et al., 1976). This is in the same range as the 800-1,500 g per day in the Minamata inhabitants poisoned by fish (Takeuchi and Eto, 1975). The health consequences depend on the dose of methyl mercury and the amount absorbed over a long period of time. As shown in Table 1, even exposure to 0.2 mg/kg of fish can result in very high tissue mercury levels in humans if fish consumption is very high.

A recent study has addressed the relationship between mercury in fish from reservoirs and mercury in human hair for human populations in northern Finland (Lodenius et al., 1983). Fish in man-made lakes had higher mercury than those in a natural lake with "background" levels, and there was an associated increase in mercury in human hair for people living near the man-made lakes. Within the exposed populations, middle aged people eating substantial amounts of fish from reservoirs contained the highest (30 mg/kg) hair mercury compared with a background mean of 1.6 and a maximum of 6.0 mg/kg in hair. Symptoms of Minimata disease may occur when hair levels of mercury are 50 mg/kg or more (Piotrowski and Inskipp, 1981). A Swedish expert committee considered that mercury in hair should not exceed 6 mg/kg.

Dose-response relationships for methyl mercury poisoning are generally derived from data obtained during acute outbreaks of mercury poisoning, using graphical extrapolation of an assumed linear relationship between

body burden and symptoms to zero incidence, of the symptom and the associated minimum body content (Bakir et al., 1973). Data from a population of Indians in northwestern Quebec, who were exposed to mercury contaminated fish but who showed no overt symptoms of mercury toxicity were analyzed by Barbeau et al. (1976). Blood mercury levels were used to reflect intake of methyl mercury and specific neurological and behavioural functions were documented. Based on blood levels, it was calculated that individuals consuming 320 g (11.3 oz) per day of fish containing 0.5 mg/kg mercury would develop symptoms of methyl mercury toxicity over a typical summer season when fish consumption was high.

The present marketing standard for Canada is a permissible level of 0.5 mg/kg or less in fish (Rose, 1979). The U.S., Japan, Sweden and WHO standards are currently 1.0 mg/kg. This provides a large margin of safety for populations on a diverse diet (Table 1), and is considered sufficient to protect even those individuals whose diet is monotonous and high in fish. There are many unknowns, however, including the effect of nutritional status of the exposed population. For example, thiamin and magnesium deficiencies may increase the vulnerability of individuals to methyl mercury poisoning, or may lead directly (i.e. without involving mercury) to symptoms similar to those attributed to mercury toxicity.

Recognizing the considerable uncertainties in any evaluation of the risk to human health of mercury as an environmental contaminant, there are nevertheless sufficient examples of fish from remote lakes with mercury levels exceeding the permissible level to merit concern at the practical level and also to stimulate studies on the fundamental mechanisms underlying fish mercury accumulation.

2.5 MERCURY UPTAKE BY FISH

Fish can accumulate mercury either through food or directly from the water. In natural ecosystems, it is most likely that fish accumulate mercury via these two vectors concurrently. However, the relative importance of these two uptake processes has not been clearly established,

even under laboratory conditions (Huckabee et al., 1979), and is probably related to species and site-specific conditions.

The accumulation of mercury from food must be related to concentration in the food, amount of food ingested, and efficiency of assimilation across the gut wall. All of these factors are highly species specific. For example, a 3.0 kg northern pike (Esox lucius) will consume a greater amount of food than a 0.25 kg rainbow smelt (Osmerus mordax). Furthermore, the mercury content in the diet of the piscivorous pike will be higher than in that of the largely planktivorous smelt. This fact is due primarily to the process of biomagnification within the food chain. The concept of biomagnification between fish species has been criticized as an oversimplification of mercury transfer within aquatic ecosystems (Sharpe et al., 1977; Huckabee et al., 1979). Critics claim that observational data are often misleading because the age factor of fish is not accounted for, and that predatory species are often much older than prey species. However, recent studies of freshwater food chains taking the age factor into consideration clearly demonstrate that biomagnification of mercury does exist (MacCrimmon et al., 1983; Wren et al., 1983).

The efficiency of mercury assimilation from food appears to vary among species. Experimental studies have shown that 68-80% of mercury ingested is assimilated by rainbow trout (Phillips and Buhler, 1978; Rodgers and Beamish, 1981), compared to 20% in northern pike (Phillips and Gregory, 1979); an efficiency of 80% has been used in uptake models for yellow perch (Norstrom et al., 1976). However, it has also been shown that mercury assimilation rate across the gut will begin to decrease above certain critical concentrations (Rodgers and Beamish, 1981).

The amount of mercury accumulated directly from the water by fish is more difficult to calculate due to problems in accurately measuring mercury concentrations in surface waters. Mercury uptake from the water will be determined by the water concentration, fish metabolic rate and efficiency of uptake (bioavailability) as determined by the ambient water characteristics. The latter factor is the least understood, and perhaps the

most important condition governing mercury uptake in fish living under natural conditions.

A number of detailed models have been developed relating mercury uptake in fish to fish metabolic rate (Norstrom et al., 1976; Rodgers and Beamish, 1981). Several biotic and abiotic variables are directly related to fish metabolic rate and, therefore, it is not surprising that these variables also control the rate of mercury uptake from water. These variables include water temperature (MacLeod and Pessah, 1973; Reinhard et al., 1974) and fish body size (DeFreitas and Hart, 1975; Sharpe et al., 1977). Rodgers and Beamish (1981) showed that uptake of waterborne methyl mercury by rainbow trout was linearly related to oxygen consumption and methyl mercury concentration at values less than 8.0 ug/L.

The uptake efficiency of methyl mercury by fish from water has been estimated at 7.1% (Boddington et al., 1979) and 8.3% (Rodgers and Beamish, 1981) in salmonids. The latter value was obtained while running the experiment in hard water (385 mg/L $CaCO_3$). When the experiment was repeated in soft water (30 mg/L $CaCO_3$), the uptake efficiency was increased to 25% (Rodgers, 1982). Increased uptake efficiency may therefore partially explain elevated mercury levels in fish from low pH lakes. The rate of accumulation and toxicity of several metals to fish are known to be reduced by increased water hardness or calcium levels (Pickering, 1974; Davies et al., 1976; Howarth and Sprague, 1978; Carrol et al., 1979). The mechanism to account for this effect is thought to be increased gill permeability at low calcium levels (Spry et al., 1981), or competition between heavy metals and calcium for cellular binding sites (Zitko and Carson, 1976). Observations from field studies also show correlations between fish mercury level and water calcium concentration (MacFarlane and Franzin, 1980; Wren and MacCrimmon, 1983).

A variety of other physico-chemical conditions within lakes have been empirically related to mercury levels in fish. Prominent among these relationships are elevated fish mercury levels at lower lake pH (Suns et al., 1980; Jernelov et al., 1975; Brouzes et al., 1977; Hakanson, 1980;

Wren and MacCrimmon, 1983). Sloan and Schofield (1983) reported that mercury levels were higher in brook trout from drainage lakes (pH $\leq$ 5.0) than in brook trout from seepage and bog-type lakes (pH > 5.0). However, they also state that trout mercury levels did not change after liming one lake, and this conflicts with the pattern of other data presented. A recent evaluation of data from over 200 lakes in Sweden suggests that mercury levels are elevated in fish within specific regions due to increased atmospheric deposition of mercury, and also lake acidification (Bjorklund et al., 1984). Despite these studies, no plausible mechanism to account for the effect of water pH on fish mercury levels has yet been put forth. The original hypothesis concerning the production of monomethyl mercury at pH < 6.0 (e.g. Beijer and Jernelov, 1979) is almost certainly a gross oversimplification of the processes occurring within natural ecosystems. The recent work of Xun et al. (1985) (see Section 2.3 above) on the effects of pH on methylation and demethylation rates emphasize the complexity of this issue.

2.6 MERCURY IN FISH FROM RESERVOIRS

The earliest documentation of mercury in fish from impoundments appeared when Smith et al. (1974) reported on high mercury levels in fish from Willard Bay Reservoir, Utah. This is a shallow reservoir with a pH of 8.1-8.3. As shown in Table 2, predatory fish in particular contained high total mercury (from 0.16-7.3 mg/kg). However, only two of the fish species from Willard were common to other Utah water bodies reported by the same authors, and the authors at that time provided some evidence that Willard was in fact 'naturally' polluted with mercury. They stated that "a major contribution of mercury to Willard Bay Reservoir may be made from insoluble mercury salts or sulfides in the muds deposited from the time of Lake Bonneville which receded to form the highly saline Great Salt Lake". The only geochemical measurements referred to were those of the 1971 geological survey where "The greatest concentration of mercury found in a soil sample was from Summit County, Utah". Without any more specific data it is not possible to speculate further on the significance

Table 2

MERCURY LEVELS IN VARIOUS FISH SPECIES FOR RESERVOIR STUDIES IN THE U.S.A. (ALL VALUES IN mg/kg WET WEIGHT)

	Carp	Walleye	Largemouth bass	Brown Trout	Reference
Willard Bay Reservoir, Utah	0.52 - 1.210	0.160 - 3.890	0.270 - 7.300	1.33 (1 sample)	Smith et al., 1974
Reference area, Utah	0.040 - 0.820	- - - -	- - - -	0.130 - 0.430	Smith et al., 1974
Lake Powell, N. Mexico	0.181 - 0.339	0.209 - 0.763	0.192 - 0.688	0.073 - 0.101	Potter et al., 1975

of the observations on fish mercury; the study is included here as an example of the high mercury levels attained in predatory fish in reservoirs.

More comprehensive are the results of a study on mercury in a desert reservoir in New Mexico (Lake Powell), reported by Potter et al. (1975). Flooding commenced in 1963 and by 1971 the volume of the reservoir was 17,860 x 10^6 m^3. Potter et al. (1975) measured mercury in seven different fish species, for eleven different tissues. Ranges for four of the species are given in Table 2, for axial muscle (which was the tissue highest in mercury for most samples). Predatory fish commonly had mercury > 0.5 mg/kg but none exceeded 1.0 mg/kg. Fish mercury was related to body weight and trophic level. There are no pre-impoundment data nor reference areas, and as shown in Table 2, the mercury levels in fish, where there are species in common, were generally much lower than for the Utah study of Smith et al. (1974). Nevertheless the Lake Powell study is valuable because the authors present a mercury budget for the reservoir, which indicates that apart from fish, the highest mercury levels were in "water transported" plant debris, which contained up to 0.145 mg/kg (dry weight). This was significantly higher than for live plants in the terrestrial and aquatic systems, which never exceeded 0.034 mg/kg in leaves. The authors gave evidence for 'bioamplification' of mercury, and considered that in this reservoir, remote from pollution sources and with "only very slight and very local pollution", mercury was released by "natural weathering" of the basin. Soil samples did not indicate unusually high mercury levels, with the highest values (.01-.05 mg/kg) being found in the relatively scarce substrates containing appreciable levels of organic matter. They calculated that 800 kg/yr might accumulate in the reservoir.

Abernathy and Cumbie (1977) reported on mercury accumulation in largemouth bass for three reservoirs in South Carolina. The waters contained mercury levels below the detection limit of 0.06 ug/L. These authors showed relatively high fish mercury with differences among reservoirs of different ages (Table 3). The highest fish mercury was in the young,

Table 3

MERCURY CONCENTRATIONS (mg/kg WET WEIGHT)
IN BASS FROM A NUMBER OF RESERVOIR STUDIES

Reservoir	Mercury Concentration (mg/kg)	References
L. Jocassee S. Carolina (1 year)	1.87 - 4.49	Abernathy and Cumbie, 1977
L. Hartwell S. Carolina (11 years)	0.38 - 0.68	Abernathy and Cumbie, 1977
L. Keowee S. Carolina (3 years)	0.34 - 3.99	Abernathy and Cumbie, 1977
Cedar Lake Illinois	0.07 - 0.50	Meister et al., 1979
Ross Barnett Mississippi	<0.05 - 0.74	Knight and Herring, 1972
Occoquan Res. Virginia	0.00 - 0.8	Gawlik, 1979
Willard Bay Utah	0.270 - 7.3	Smith et al., 1974
Lake Powell New Mexico	0.192 - 0.688	Potter et al., 1975

Table 4

SUMMARY OF AVAILABLE RESERVOIR STUDIES

Water body	Location	References	Hg in fish	Fish spp. or age given	Pre-impound-ment data; or refer-ence area given	Post-impound-ment or time trends	Hg in water or sediments	Water quality data	Hydrol/ Physical informa-tion	Other Data	Comments
S. Indian L	Man.	Bodaly and Hecky, 1979	x	x	x	x	x	x	x	Bank eroded Hg in moss, soils etc.	Experimental + Se data
		Bodaly et al., 1984b	x	x	x	x	x	x	x		Reference to natural lakes in area where fish Hg not increased
		McGregor, 1980	x	x	x	x					4 fish spp.
		Env. Canada, 1979	x								
Rat L.	Man.	McGregor, 1980	x	x	x	x					4 fish spp. Rat L. fish Hg>SIL
		Bodaly et al., 1984b	x	x	x	x			x		
Notigi L.	Man.	McGregor, 1980	x	x	x	x				Hg in moss soils etc.	4 fish spp. Notigi fish Hg>SIL
		Bodaly et al., 1984b	x	x	x	x			x		
Wuskwatim	Man.	McGregor, 1980	x	x	x	x			x		4 fish spp. Wuskwatim fish Hg>SIL
		Bodaly et al., 1984b	x	x	x	x			x		
Smallwood R.	Labrador	Bruce and Spencer, 1979	x	x	x						14 fish spp. Hg levels highest in piscivores
Cookson	Sask.	Waite et al., 1980	x	x			x			Ref to Expt (IJC) work	Hg highest in walleye, water Hg low
Strutt L.	N.W.T.	Weagle and Cameron, 1974 in Bodaly and Hecky, 1982	x	x							Abstract only

Table 4
(cont'd.)

Elephant Butte R.	N. Mexico	Garcia and Kidd, 1979				x	x	x		Great fluctuation in water level; difference in water Hg across reservoir
Lahonatan R.	Nevada	Cooper, 1983	x	x				x	Crayfish & Seagull Hg	Impounded 1915. Weight/Hg correlation for some (11 spp. examined) Source of Hg is old mining.
		Cooper et al., 1983	x	x		x	x	x		Reservoir is eutrophic. Carp mortality in 1980 and 1981
B. Everett Joshia R.	N. Carolina	Shuman et al., 1983	x		x	x		x	Size-fraction of Sediment had different Hg content	Municipal discharge contains Hg, and natural source
General	Canada	Paxter and Glaude, 1980								Has list of reservoirs across Canada
Various (80 sites)	Illinois	Hullinger, 1975				x				Map of locations in Illinois. High water Hg seems to be in certain regions only small sample number. Analytical methods may be suspect.
Boundary Dam	Saskatchewan	Allan and Richards, 1978			x	x	x			Original water quality poor, surface sediment high in Hg (as for other metals). TDS downstream improved after dam built.
Liard River Drainage (pre impound-ment)	B.C. Canada	Sekerak and Mace, 1982	x		x		x			Metals low in water, sediment and fish
La Grande 2 and Opinaca	Quebec James Bay	Boucher and Schetagne, 1983	x	x	x	x	x			Lots of raw data and good time trends. Fish Hg increased years 1-2 and more in year 4. Erosion neglibible, pH low.

oligotrophic reservoir and the lowest in the older, eutrophic reservoir. The results do not conclusively relate fish mercury to impoundment or to reservoir age. There were other large differences among reservoirs, in water quality parameters such as conductivity, alkalinity, pH and turbidity, all of which could affect mercury availability. The authors suggested that anaerobic conditions in the more nutrient-rich systems would result in binding of mercury to sulphur compounds and organics, which would remove the mercury from the food web. They refer briefly to methylation, which they consider would be favoured by aerobic conditions prevailing in the more oligotrophic systems, and conclude that the source of mercury to fish "could easily have been provided by background levels of mercury which were present in topsoil which formed the sediments of the reservoirs after filling".

In a follow-up study for Lake Jocassee, which had the highest fish mercury, Abernathy and Cumbie (1977) found a decrease between 1974 (1.9± 0.12 mg/kg) and 1975 (0.69 ± 0.06 mg/kg). Based on this and on the data for different aged systems, the authors concluded that "elevated mercury in fish of new reservoirs is probably transitory, declining within 3-5 years after reservoir filling under conditions which exist in the systems we have studied".

Meister et al. (1979) also identified soil from the flooded area which sedimented into the reservoir as the original source of mercury for crappie and largemouth bass in Cedar Lake Reservoir, Illinois. Some of the heaviest (3.5-5.5 kg) bass contained more than 0.5 mg/kg mercury (Table 3). Meister et al. considered that microbial methylation of the sediment mercury, combined with food chain biomagnification, resulted in elevated fish mercury particularly in predatory fish, and that "anaerobic conditions that favour the uptake of mercury from soil by methylation can occur at the lake bottom". They concluded that "elevated mercury in predatory fish in new impoundments is to be expected and is independent of water quality". Furthermore, they expected that eventually the concentration should drop because of constant removal of water, fish harvest and volatility of organic mercury. Cox et al. (1979) in another report

on Cedar Lake considered that the source of mercury was natural and cautioned that if reservoirs are to be used as fisheries, close monitoring of mercury in predatory fish should be carried out during the first few years.

A number of other studies from the U.S. and Canada have been published during the late 1970s and early 1980s. In order to give the reader a sense of the extent of the studies and their respective limitations, these are summarized in Table 4. Even when one excludes those situations where a specific source of mercury is identified, e.g. mining activity (Phillips et al., 1980; Cooper, 1983) or contamination from a chloralkali plant (Hildebrand et al., 1980), it is clear that fish mercury is frequently unacceptably high in impoundments remote from major sources and that the phenomenon has been described for a variety of regions, from east to west, of the North American continent. Whether there are conspicuous exceptions remains to be seen; only rarely do studies report on data which do not indicate unexpectedly high fish mercury levels. Furthermore, many of the earlier studies were commenced after flooding, and time trends or pre-impoundment data are not available. While the phenomenon of elevated mercury in impounded waters is widely viewed as the rule rather than the exception, very few studies (Boucher and Schetagne, 1983; Bodaly et al., 1984b) actually provide direct evidence of changes in mercury levels after impoundment.

Within Canada, several recent studies which are fairly complete can be highlighted as examples of the geographical extent of the phenomenon. In Labrador, preliminary studies in 1976 (Bruce and Spencer, 1979) for Smallwood Reservoir which was filled in 1971 showed "above background" levels of mercury in several fish species. A comparison of mercury in the different fish species from 88 locations in Labrador showed that the highest mercury levels were in fish from Smallwood Reservoir and the forebay downstream of the Lobstick Control structure. The highest levels were for lake trout (3.9 mg/kg) and burbot (1.93 mg/kg) from the reservoir. Thus although no pre-impoundment data were available, spatial comparisons indicated abnormal levels in the reservoir fish.

Table 4
(cont'd.)

Various	N. Finland	Lodenius et al., 1983	x	x						Values for human hair by region	Increased Hg in reservoir & more in top predators. Highest hair Hg in people eating reservoir fish.
Cedar L.	Illinois	Meister et al., 1979	x	x	x		x			Small number of centrarchids	Soil probable origin of Hg
		Cox et al., 1979	x	x	x						
L. Powell	Arizona	Potter et al., 1975	x	x		x		x		Hg in terrestrial plants, aquatic plants, invertebrates, & fish tissues	
Tongue R.	Montana	Phillips et al., 1980	x	x							Sites contaminated from mining
Hartwell, Keowee & Jocassee	S. Carolina	Abernathy and Cumbie, 1977	x	x			x	x			High levels in fish from reservoir; diffs not clearly related to impoundment Jocassee & Keowee have highest fish Hg
Major Reservoirs(9)	S. Carolina	Abernathy, 1979	x				x			Calculations on soil as Hg source	
Willard Bay R.	Utah	Smith et al., 1974	x	x	x						75% fish in reservoir > 0.55 mg/kg. Reference sites much lower
Pueblo R.	Colorado	Hermann and Mahan, 1977	x								No biological data
American Falls R.	Idaho	Kent and Johnson, 1979	x	x	x		x				Very high Hg in water. Methods not given.
		Johnson et al., 1978	x			x	x	x	x	Drawdown 1976 Some experiments on sed release of Hg with temp. & DO	Follow up of 1974 paper. Water Hg still high

Table 4
(cont'd.)

Cherokee Reservoir	Virginia & Tenessee	Hildebrand et al., 1980							Chloralkali plant was source of Hg
B Everett Jordan Falls L.	N. Carolina	Weiss and Bond, 1978			x	x	x	Concern was for water content	Over 140-150 days, Hg increased, decreased later. Suggest [Hg] associated with richness of organic content.
B Everett Jordan Falls L.	N. Carolina	Shuman et al., 1983	x					Sediment metals. One arm of system had most contaminated fish.	Goldmining area. 1032 sports fish collected; 28% had Hg > 0.5 mg/kg.
Occaquan Reserve	Virgina	Gawlik, 1979	x	x				Hg highest in L.M.bass	English reservoir; Agricultural and urban run off

In Quebec, reservoirs constructed on the La Grande River were studied over the period 1978-82 (Boucher and Schetagne, 1983). These authors compared mercury in fish for pre-impoundment with post-impoundment conditions. At all sites, mercury was consistently higher in the piscivorous pike and walleye. For all fish species there was a correlation between age and mercury, or between length and mercury, but there was a great deal of variability in the data. After impoundment, mercury in fish increased: e.g. walleye year 2 (2x) and year 4 (3.5x), and for whitefish in year 2 (3x) and in year 4 (5.5x). Table 5 summarizes some of their results. Most fish in the reservoirs exceeded the federal 0.5 mg/kg limit, and it was recommended that fish consumption should be limited.

In a discussion of the possible mechanisms, Boucher and Schetagne (1983) considered the role of sediment additions resulting from erosion, and of organic matter, as factors mediating the influx and methylation rates of mercury. For La Grande Reservoir, however, erosion had been negligible and no increase of organic matter in solution or suspension had been observed. These authors considered that a major controlling factor for methylation in the specific systems under consideration was a decrease in pH of surface waters from about 6.5 to 6.0, with slightly lower (5.4-5.9) pH in subsurface water. Inundation of vegetation was also considered to be an important factor, since terrestrial plants, especially mosses, have a high metal binding capacity, and may be loaded with mercury (originally from the atmosphere). A third factor which these authors discussed was the concentration of nutrients, which could enhance methylation of mercury. After flooding, La Grande and Opinaca reservoirs did in fact have higher concentrations of nutrients than before.

The Cookson Reservoir in Saskatchewan was created in 1976 in an area remote from anthropogenic sources of mercury. Data for mercury were collected in 1979 (Waite et al., 1980). Walleye in the reservoir exceeded the 0.5 mg/kg limit, but there were no pre-impoundment or reference site data. For this study, there were insufficient data on other factors to permit speculation concerning the mechanism, and "preliminary information on mercury concentrations in terrestrial soils and reservoir sediments do

0 0 3 2 9

Table 5

PRE-AND POST-IMPOUNDMENT VALUES FOR
FISH MERCURY LEVEL IN QUEBEC RESERVOIRS

Reservoir	Fish Species	Pre-impoundment levels	Post-impoundment Year2	Year 4
La Grande 2	Lake whitefish	0.09 - 0.17	0.26	0.51
Opinaca	Longnose sucker	0.13 - 0.19		0.40
Caniapiscau	Pike	0.52 - 0.60		1.24
La Grande 2 Opinaca	Walleye	0.60	1.28	2.05
Caniapiscau	Lake Trout	0.88		
Rivers				
Nottaway	Grand coregone	0.18		
Broadback	Walleye	0.58		
Rupert	Pike	0.60		

All concentrations mg/kg wet weight

From: Boucher and Schetagne, 1983

not show unusually high levels" but they state that "if the mercury source is newly inundated soil containing average southern Saskatchewan mercury concentrations and if there are no other sources, it is likely that mercury concentration in the fish of Cookson Reservoir will peak and then begin to decline".

The most comprehensive studies to date regarding the mercury-reservoir problem come from a hydroelectric development in the Churchill and Nelson river valleys in Northern Manitoba. This is an immense development, in which the combined flow of two of Canada's major rivers is utilized to provide hydroelectric power to Manitoba. Eight hundred twenty m^3/s of the 960 m^3/s mean annual discharge of the Churchill River is now diverted into the Nelson River basin (Bodaly et al., 1984a). Prior to development, the ecosystems were riverine lakes (10-2000 km^2) on the precambrian shield, with bedrock overlain by glacio-lacustrine and glacial deposits of clay, silt and sand. The climate is subarctic and much of the soil is permanently frozen. The diversion began in 1976, and details of the works and hydrology are described by Newbury et al. (1984). In 1976, the level of Southern Indian Lake (SIL) was raised by 3 m which flooded 414 km^2.

Pre- and post-impoundment studies on a number of ecological and physical aspects have been conducted at SIL and have been described in a recent series of papers (Newbury and McCullough, 1984; Hecky and McCullough, 1984; Hecky, 1984; Hecky and Guildford, 1984; Bodaly et al., 1984b). Based on the reservoir paradigm (Rzoska, 1966) a number of predictions of physical changes were correct, but for the most part biological changes were not correctly predicted. In particular, increases of mercury concentrations were unpredicted. Bodaly et al. (1984b) provide data on pre- and post-impoundment levels for whitefish, pike, and walleye, from SIL and from several other lakes in the same area, also affected by the impoundment. These data clearly indicate post-impoundment increases in mercury levels. It is noteworthy that this was the first system for which a change in fish mercury levels with impoundment had been documented with a time series of data (Bodaly and Hecky, 1979; Bodaly et al.,

1984b). Reference is made to spatial comparisons, in that data for fish mercury were available for 30 other lakes in northern Manitoba and during the 1970's these did not show any trends to increasing mercury. Bodaly et al. reject the hypothesis that increased atmospheric loading could account for the sudden surge in fish mercury beginning in 1976, although according to lake sediments there has been a very slow increase in mercury deposition, with an approximate doubling in flux to sediments in this and other pristine areas since 1900. It is also convincingly argued that anthropogenic point sources could not explain the surge, and mercury-rich geologic formations are also unlikely to be the cause.

Analysis for mercury in vegetation litter, other soil horizons, lake and suspended sediment and water are given, and none of these is high enough to be considered contaminated; rather they tend to be below average compared with similar materials elsewhere. Bodaly et al. (1984b) consider that the activity of flooding soils resulted in increased bacterial methylation of naturally-occurring mercury, with the primary source of mercury in the organic rich upper soil horizon. The study of Furutani and Rudd (1980) supports the idea that increased bacterial methylation would occur when terrestrial vegetation, moss, peat and humus are inundated. In a more recent study, Ramlal et al. (1985b) measured ratios of specific methylation to demethylation rates for sediments in a recently flooded reservoir. They found the highest ratios in the shoreline areas (i.e. where inundation had recently occurred) and related this to observed increases of methyl mercury in fish which occurred after flooding.

2.7 MECHANISMS

There is little disagreement that regardless of the situation, high levels of mercury in fish result from methyl mercury (e.g. Ribeyre and Boudou, 1983). This assumption can be applied to highly polluted situations such as the English Wabigoon River System (Bishop and Neary, 1975) as well as to lakes and rivers in remote areas which have not been impacted by man (Lindqvist et al., 1984), and reservoirs which although

impacted by man have no known anthropogenic source of mercury. There is also general agreement that 70-80% of the mercury in fish muscle is methyl mercury (Westoo, 1966; Lockhart et al., 1972). The explanation for the now widely recognized phenomenon of elevated fish mercury in recent impoundments can therefore be narrowed down to an explanation for mechanisms of increases in methyl mercury in the systems of interest.

Where mercury in the environmental compartments (e.g. soils, sediments, water, vegetation) of the reservoirs and their adjacent environment has been measured, there is rarely any evidence of contamination or above background concentrations of mercury (exceptions to this have been noted in Table 4). Abernathy (1979) in a rough calculation provided evidence that there was sufficient mercury in the flooded soils which formed the sediments of the reservoir to account for the fish mercury. It is therefore reasonable to assume that the source of mercury is either the materials already in the lake or river prior to flooding or the flooded soils and vegetation. The mercury in these compartments is either naturally occurring, or at least at background if not pre-industrial levels (Lindqvist et al., 1984). The question of mechanism then becomes one of mobilization and methylation of existing mercury resulting from the factors which change between pre- and post-impoundment.

From the studies which have been reviewed in the present paper, it is possible to reach a consensus that soils and vegetation in the flooded areas are the source of mercury. The Churchill River studies show a relationship between the area flooded and the mercury levels in predatory fish (Bodaly et al., 1984b). There is also strong evidence that methylation is promoted by decaying vegetation or other organic matter in the flooded area. Methylation may occur in the inundated soil and vegetation, or in the material which is washed into the reservoir and becomes bed sediment or suspended particulate matter. Table 6 summarizes some of the more emphatic opinions from the papers reviewed here.

Experimental studies are now underway at the Churchill River diversion site, using large enclosures (mesocosms or limnocorrals) into which addi-

Table 6

SUGGESTED MECHANISMS FOR INCREASED
MERCURY IN FISH FROM RECENT IMPOUNDMENTS

Reservoir(s)	Proposed Mechanism	Reference
Lake Jocassee, South Carolina	Methylation favoured by aerobic conditions in the more oligotrophic reservoir. Source of mercury: background levels present in topsoil which formed sediments.	Abernathy and Cumbie, 1977
Cedar L. Reservoir, Illinois	Methylation favoured by anaerobic conditions. Source of mercury: soil which formed lake bottom.	Meister et al., 1979
La Grande river system, Quebec	Inundation of vegetation, decrease in pH, and increased nutrients in water after flooding.	Boucher and Schetagne, 1983
Churchill & Nelson river valley, N. Manitoba	Increased bacterial methylation in soils after flooding. Mercury source: organic layers of soils and flooded vegetation such as moss.	Bodaly et al., 1984b

tions of the various types of vegetation in the flooded area are made, and fish mercury levels are measured. Preliminary results (unpublished) indicate that the best promoter of methylation is Sphagnum moss. These studies are consistent with the results of other experimental studies which have shown that addition of organic substrates increased rates of microbial mercury methylation (Furutani and Rudd, 1980). A recent review of the Swedish mercury problem (Lindqvist et al., 1984) which did not in fact consider the reservoir phenomenon, but rather considered remote lakes and the related fish mercury problem, concluded that a major source of mercury is from surrounding soils and vegetation. In southern Sweden these are already contaminated by increased atmospheric fallout of mercury from long range transport. If this is so for unflooded systems, then flooding would clearly further enhance this tendency. There is also some suggestion from Swedish studies (Hultberg, personal communication) and from Japanese studies (Nagase et al., 1982) that non-microbial methylation (probably via the free radical reaction described earlier) could occur in these systems although these results were obtained with very high concentrations of mercury (1 ug/L). Non-microbial mercury methylation in experiments was stimulated by organic matter and certain metals such as Fe, Mn, and Cu (Hultberg, personal communication).

3 SUMMARY OF WORKSHOP 1

3.1 INTRODUCTION

A prerequisite to achieving the objectives of this project (see Section 1.1) is a synthesis of the current understanding of mercury problems in Canadian hydroelectric reservoirs. To this end, a workshop was held April 9-11, 1984 at the Institute for Environmental Studies, University of Toronto. The workshop was attended by representatives of hydroelectric utilities from across Canada and experts familiar with particular aspects of the mercury problem, as well as members of the study team from ESSA Ltd., LGL Ltd., and the University of Toronto (Table 7).

To establish a common focus for discussion, the participants first developed an explicit definition of the problem to be addressed by the workshop. It was agreed that the "mercury problem" concerned hydroelectric reservoirs where incremental increases in fish mercury concentration (as a result of reservoir construction) create a real or perceived health hazard to humans consuming fish from those reservoirs.

The objectives of the workshop, then, were to:

a) develop a conceptual model of the processes by which fish become contaminated with mercury following construction of hydroelectric reservoirs; and

b) evaluate the hypotheses which comprise the conceptual model for the purpose of documenting:

- evidence supporting or refuting the hypotheses;
- the value of testing the hypotheses;
- alternative methods of testing the hypotheses; and
- where feasible (i.e. within the time and financial constraints of the CEA project), develop a preliminary research plan to address identified gaps in current

Table 7

PARTICIPANTS AT THE FIRST MERCURY WORKSHOP

Name	Affiliation
Hugh Bain	Department of Fisheries and Oceans
Drew Bodaly	Freshwater Institute
Gina Cunningham	ESSA Ltd.
Fred Debidin	Ontario Hydro
Dirk deGraaf	LGL Ltd.
Lorne Greig	ESSA Ltd.
Gaetan Guertin	Hydro Quebec
Andrew Hamilton	International Joint Commission
Mike Jones	ESSA Ltd.
Dave Kiell	Newfoundland and Labrador Hydro
Gail Krantzberg	Institute for Environmental Studies
Greg McCullough	Freshwater Institute
Don McKay	Institute for Environmental Studies
Lorna McKerness	Canada-Manitoba Mercury Agreement
David Marmorek	ESSA Ltd.
Don Mortimer	National Research Council of Canada
Bill Peel	Alberta Power Ltd.
Barb Reuber	Institute for Environmental Studies
John Rudd	Freshwater Institute
Bob Ruggles	Ontario Hydro
Aaron Sekarek	LGL Ltd.
Pam Stokes	Institute for Environmental Studies
Tom Watson	B.C. Hydro
Dennis Windsor	Manitoba Hydro
Chris Wren	Institute for Environmental Studies
Bernice Young	Health and Welfare Canada

understanding of the mercury problem.

The complete results of the workshop have already been documented in an earlier report (Marmorek et al., 1984) and will not be described in detail here. Instead, the focus of this section will be on how the discussions that took place at the workshop led to the design of the research activities that followed.

3.2 WORKSHOP RESULTS AND THE RESEARCH PLAN

Most of the discussions which took place at the workshop were devoted to the development of a fairly detailed conceptual model of the mechanisms by which mercury might accumulate in fish tissues after construction of a reservoir (workshop objective (a) above). The model that emerged from these discussions is depicted in Figures 3 and 4. On the third day of the workshop, this model was disaggregated into a set of seventeen component hypotheses, which were to be evaluated with a view towards developing a research plan.

During the discussions of the first eleven hypotheses, all of which were derived from Figure 3, it became evident that though the problem can be conceptualized into separate processes, it is extremely difficult to examine these separate processes in the field. Microbial activity and methylation rates, for example, are extremely variable both spatially and temporally. This greatly impedes attempts to experimentally demonstrate each linkage in the cause-effect chain from increased organic matter to increased concentrations of mercury in fish. Experiments conducted by the Freshwater Institute have shown that fish mercury concentrations consistently integrate (and reduce) the fluctuations inherent in lower trophic levels. Therefore, rather than proceeding to evaluate the remaining hypotheses, we instead developed a simpler conceptual model (Figure 5), which does not explicitly account for the individual processes that link site factors to mercury concentrations in fish. The participants then considered each "spoke" of the "wheel" depicted in Figure 5, in terms of existing data which might be used to assess its importance, and

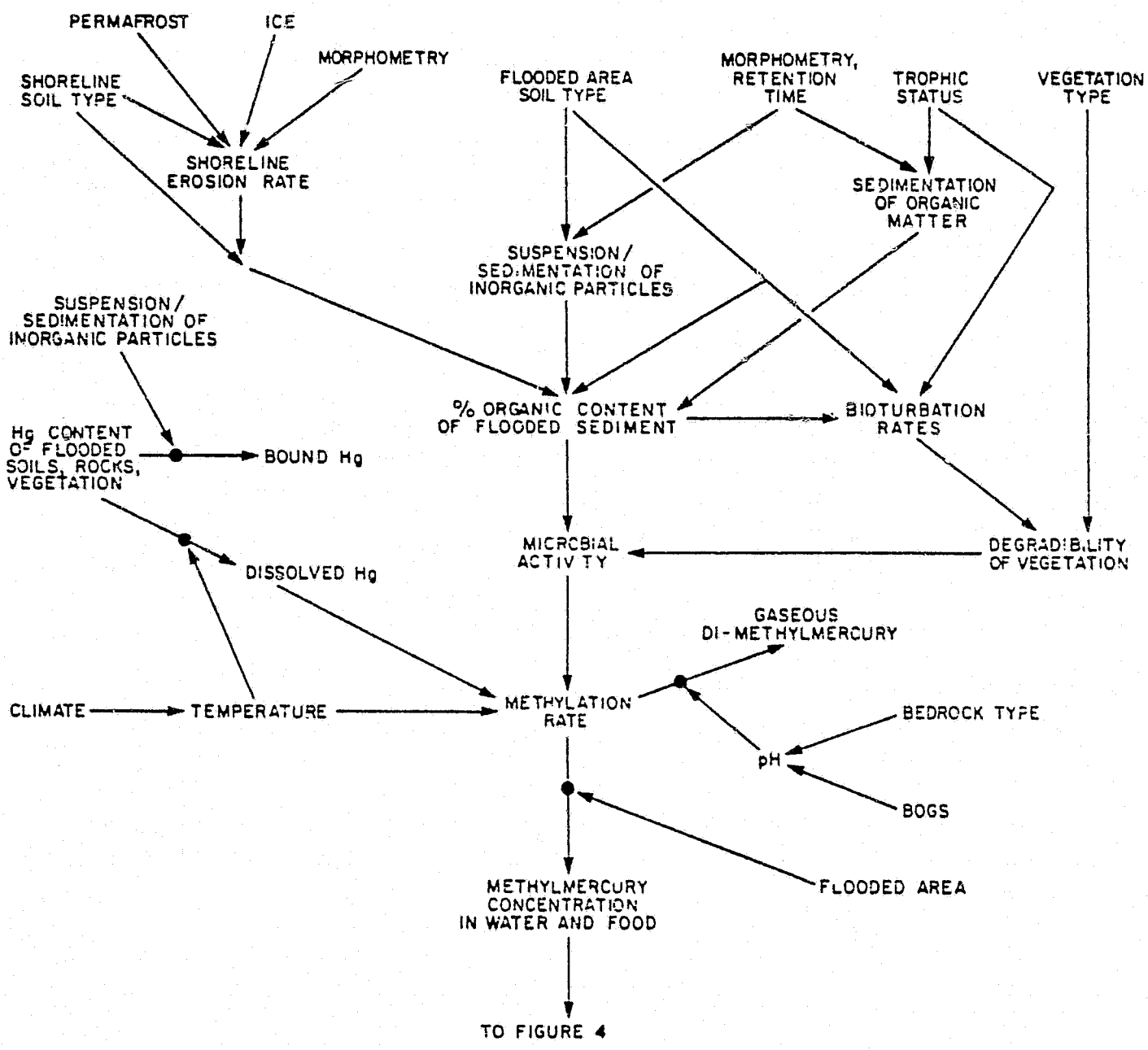


Figure 3 Conceptual model of factors influencing the concentration of methyl mercury in water and food.

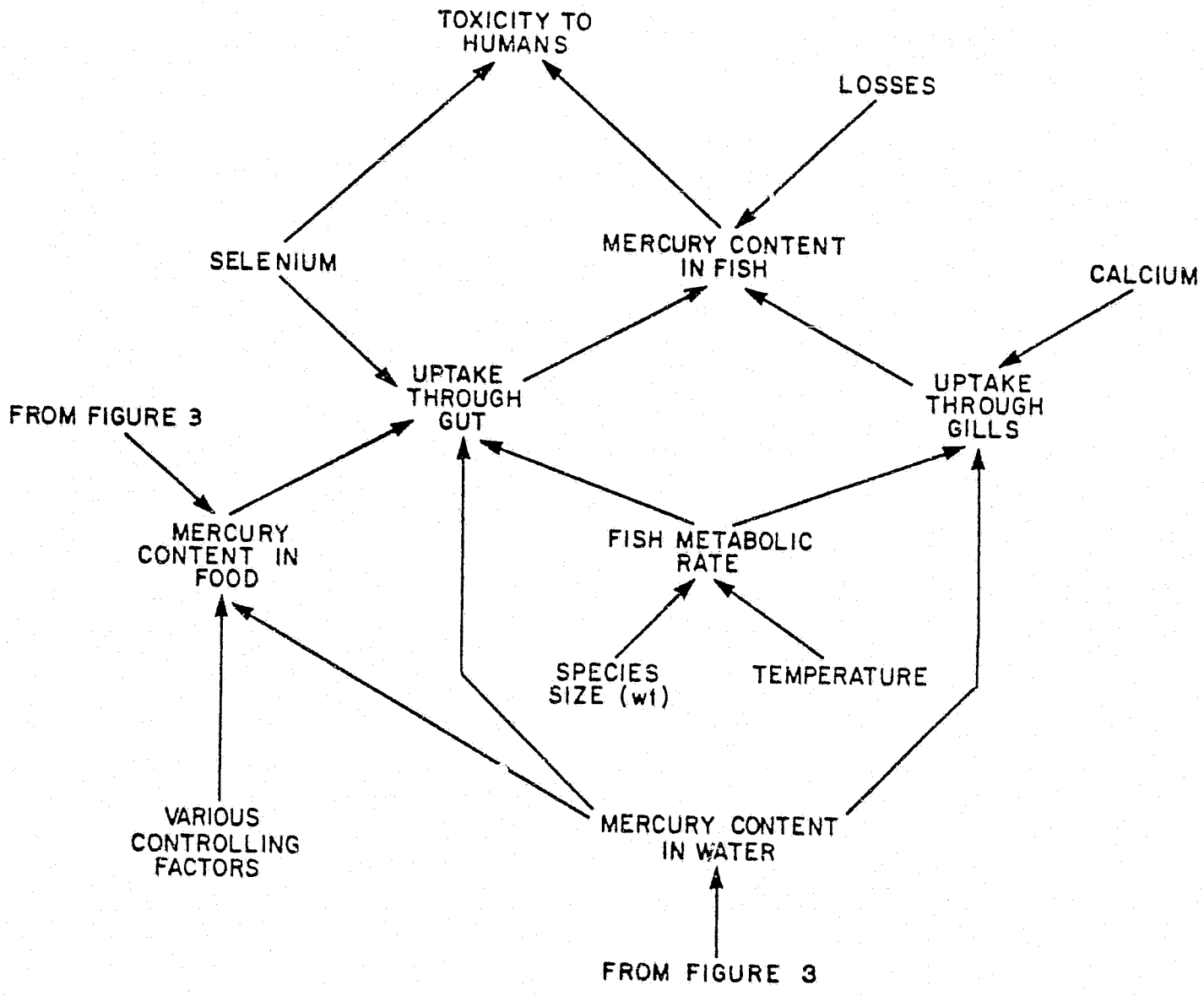
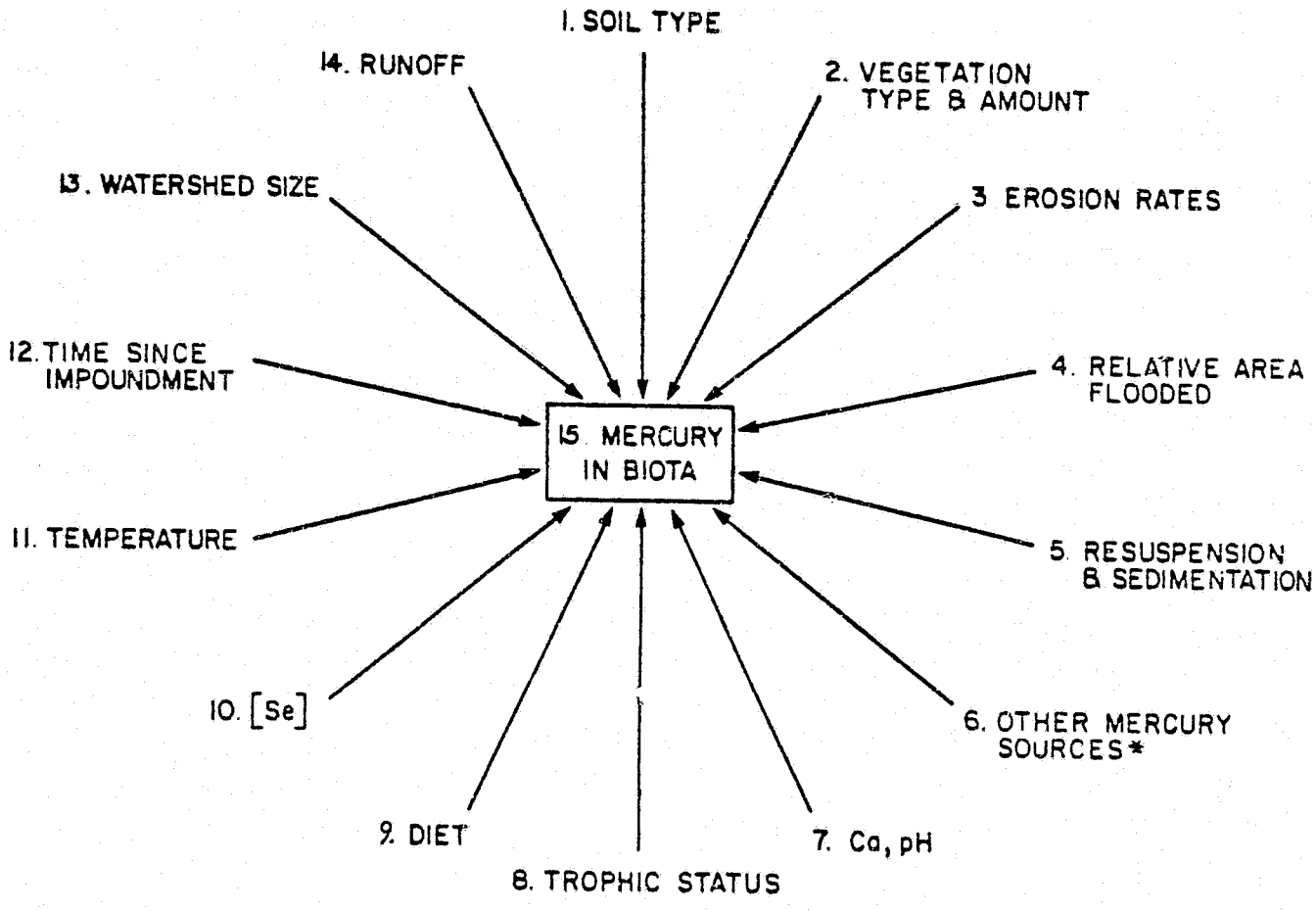


Figure 4 Conceptual model of factors influencing the mercury content of fish.



* external sources (e.g. atmospheric, upstream, industrial effluent)

Figure 5 Collapsed conceptual model of factors suggested to be linked to mercury levels in reservoir fish.

in terms of needs for further research. Note the mercury concentration of soils and vegetation is not considered as a factor in Figure 5. Discussions at the workshop, and our review of the literature (Section 2) indicate that the availability of inorganic mercury is not limiting. Rather, it is processes that influence the methylation and bioavailability of mercury which are important.

It quickly became clear from these discussions that the first priority for future research should be to compile existing data on as many Canadian reservoirs as possible for which mercury levels in fish had been measured. Data on reservoir site parameters (soil, vegetation, bathymetric characteristics) could then be used as independent variables in an attempt to explain the observed variations in fish mercury concentrations. While this approach tends to be exploratory and empirical, there was strong consensus among the workshop participants that this type of analysis would be much more valuable than a detailed field or laboratory experimental approach which focussed on specific processes, given the broad objectives and limited resources available for this study.

Our initial plan was to assemble and analyze all existing data as quickly as possible and then design a field program to collect data which might further elucidate relationships suggested by this analysis. It soon became apparent, however, that it would be impossible to complete this task quickly enough to provide useful input to the design of a field program in the summer or fall of 1984. On the other hand, our assessment of existing sources of reservoir mercury data, conducted prior to the workshop, and discussions at the workshop, suggested that a field program designed to supplement existing data would be worth carrying out.

In the absence of a detailed analysis of existing data, the design of the field program was based on two major considerations:

1) hypotheses of particular interest which might be tested by sampling a specific set of hydroelectric reservoirs; and

0 0 3 2 9

2) the desirability of having a database that includes information from all regions of Canada.

Two factors which were considered by workshop participants to play an important role in determining fish mercury levels were:

1) reservoir age; and

2) the area flooded by impoundment relative to pre-impoundment area.

In Ontario, three reservoirs in the Mississagi River drainage basin, Tunnel Lake, Red Rock Lake and Aubrey Lake, were impounded in 1950, 1960 and 1969, respectively, thus providing an opportunity to assess the importance of reservoir age to levels of mercury in fish on three reservoirs in the same drainage. As well, each of these reservoirs was sampled for mercury in fish by the Ontario Ministry of the Environment in 1977, thus allowing within-reservoir contrasts at two points in time.

In Nova Scotia, the Wreck Cove hydroelectric project resulted in the creation of four reservoirs at approximately the same time (1978) but with widely differing areas flooded relative to the surface area of the water bodies on these sites prior to flooding. These reservoirs thus provided an opportunity to assess the significance of flooded area to mercury levels in fish.

Our pre-workshop assessment of existing data suggested that mercury data were available from reservoirs in Labrador, insular Newfoundland, Quebec, Manitoba, and Saskatchewan. Limited data were also available from Ontario and B.C. (for more details on this information, see Section 6). To supplement these data, we decided to seek locations for sampling in New Brunswick, Alberta and B.C., in addition to the Ontario and Nova Scotia sites already mentioned, so as to meet our goal of a fully national database. As we were only interested in reservoirs less than thirty years old (see Section 6 for discussion of these selection criteria), with no evidence of anthropogenic sources of mercury, and which afforded relatively easy access, our choice of reservoirs was made

relatively easy. In New Brunswick, no reservoirs were found to meet these criteria; however, Long Lake marginally violated the age criteria, being only 32 years old, so was thus chosen to represent this province. In Alberta, only Brazeau reservoir satisfied the criteria, while in B.C. the Arrow and Libby reservoirs were the most suitable choices. In all, eleven reservoirs were sampled in our field program (see Section 5 for a table of latitudes and longitudes of all reservoirs sampled and Section 6 for a map illustrating locations). Details of the design and results of this sampling program are described in Section 5.

Another factor which was emphasized during the workshop as having an important influence on fish mercury concentrations was the amount and type of soil and vegetation flooded as a result of impoundment. This factor has been the subject of investigations by scientists from the Freshwater Institute at SIL in Manitoba, in which experimental field enclosures have been used to assess the relative importance of different types of flooded materials.

As an alternative to the SIL limnocorral experiments, participants at the workshop suggested that laboratory microcosms might be designed to conduct similar experiments with greater replication under more economical conditions. The primary thrust of the suggested experiments would be to determine whether it is possible to correlate general characteristics (e.g. amount, type) of soil/vegetation with subsequent levels of mercury in fish. In the event this proved feasible, the techniques developed could help provide an inexpensive diagnostic tool for screening flooded materials for their methylation potential and might also prove useful in helping to assess the potential of proposed mitigative measures, without conducting "whole reservoir", or limnocorral, experiments. The design and results of the microcosm experiments are discussed in Section 4.

Before turning to the methods and results of the research activities which followed the workshop, one final point is worth making here. In our initial conception of this study, we envisaged developing a dynamic computer simulation model, based on the research conducted during the

study, and on existing information, which would allow one to simulate changes in fish mercury levels through time, under a variety of reservoir site selection and construction scenarios. It soon became apparent during the workshop that our process level understanding of mercury in newly impounded reservoirs is insufficient to justify the development of such a model as a useful evaluative tool. There are simply too many closely interacting factors, especially at the biochemical and microbiological level, about which we know too little to develop a defensible model of this type.

On the other hand, the research we chose to undertake as part of this study, does lend itself to the development of another, less detailed type of model. Instead of attempting to simulate the specific processes which influence mercury dynamics, the model is developed by assessing statistically the relative importance of various factors which are thought to affect mercury levels in fish (i.e. the "spokes" of Figure 5) and then developing a ranking scheme for these factors. Such a model is sometimes called a "hazard rating model", since it can subsequently be used to assess the "risk" or "hazard" associated with the selection of a new site for reservoir construction. In Section 7 of this report, where we present the results of our analysis of reservoir mercury data, we also discuss a preliminary hazard rating model which we have developed on the basis of these analyses and on our review of existing knowledge.

4 MICROCOSM EXPERIMENTS

Of the many factors that were discussed at the workshop as having an influence on mercury levels in fish (see Figure 5), soil and vegetation type were considered to be of particular importance. Studies conducted in field mesocosms (limnocorrals) at SIL have indicated that additions of organic matter (sphagnum moss, spruce boughs) stimulate significant increases in mercury uptake by fish (R. Hecky, personal communication). Participants at the workshop felt that substrate type was therefore the most appropriate candidate as a factor for deliberate experimental study.

Field mesocosms, while offering perhaps the most realistic alternative to whole lake (or reservoir) experiments, are very expensive to design and operate. A less costly option is to use much smaller laboratory microcosms. At the workshop there were differing views on the utility of microcosms as assay tools. There was consensus, however, that if suitably validated (perhaps by comparison with mesocosm or whole-system results), microcosms might provide a valuable, inexpensive method of assessing the relative stimulation of uptake by different substrate types. We therefore decided to carry out a set of microcosm experiments, to examine the feasibility of using them for this purpose. Our objective was to design the experiments in a manner that would facilitate comparison of the results to those obtained from mesocosm studies at SIL.

4.1 METHODS AND EXPERIMENTAL DESIGN

The microcosm experiments were designed to address two specific questions:

a) Does the presence of decaying organic matter stimulate the uptake of mercury by biota? and

b) Do uptake rates vary depending on the type of sediment present?

A preliminary set of experiments was conducted in the late summer of 1984, and used yellow perch (Perca flavescens) as the assay species. While the choice of species is relatively unimportant (since we are

simply using the fish as an indicator of mercury release from the sediments), perch were selected to facilitate comparisons with the SIL studies, where this species was also used. To examine the first question, samples of sphagnum moss were obtained from the vicinity of SIL, while sediments from SIL and from Guelph Lake, a southern Ontario reservoir, were used to address the second question.[1]

To facilitate comparison of the microcosm results to those obtained from the SIL mesocosms, one would ideally like to scale the microcosm experiments (in terms of size, time, and biological conditions) to be comparable to the larger enclosures. Assuming that the important size scaling factor is water volume (the medium for the biota of interest) relative to sediment surface area (the source of mercury), this leads to substantial logistical problems. To maintain this ratio constant (water volume: sediment surface area), the depth of water in the microcosms must be equal to that of the mesocosms, since

volume = (surface area) * (depth).

The mesocosms were 3 m deep, which is obviously an impractical dimension for a laboratory microcosm. Since we were interested in evaluating a fairly simple and inexpensive method, as well as one which lent itself to easy replication, we concluded that designing the microcosm experiments to be comparable in this respect was not appropriate for this study.

It is worth stressing that differences also existed in time scales and biological conditions between the microcosm and mesocosm experiments. Microcosm experiments were of short duration (30 days) relative to the limnocorral work (60 days); this discrepancy is further discussed in Section 8.3. With respect to biological conditions, by supplying the test fish with 'clean' food (i.e. not mercury contaminated) the food chain pathway was eliminated as a source of mercury in the microcosm experiments. In the mesocosms, perch presumably obtained food from the same

[1]SIL moss and sediments were kindly provided by Drew Bodaly, Freshwater Institute.

sources as they would in their natural environment. This may explain some inconsistencies in results from the two types of study.

A second set of experiments was conducted during the winter of 1984/85. We were unable to obtain an adequate supply of small perch at this time, and were thus forced to choose an alternate species. Since none of the species used at SIL were obtainable, and since fish were only being used as an indicator of mercury release from sediments, we saw no compelling reason to choose a species normally found in Canadian reservoirs. Instead, we opted to use a species which is easy to maintain in the laboratory, and which has been used in other investigations of mercury uptake, namely guppies (Poecilia reticulata).

It should be emphasized that from the point of view of interpreting the results, the choice of species is unimportant, so long as all species will increase their uptake of mercury, to a greater or lesser extent, when more is available. The choice of species for these experiments was thus made on the basis of facilitating comparisons with SIL (perch), or to minimize logistical difficulties (guppies).

In this second set of experiments, sphagnum moss was again used as the source of organic matter, but the amount added to each microcosm was varied. Two sediment types were also examined, in this case from Guelph Lake and from Lake Scugog, a natural lake in southern Ontario.

4.1.1 Experiment 1: Perch

As well as addressing the two questions stated above, the perch experiments were also designed to consider the effect of adding inorganic mercury to sediments. Such additions have often been used in similar experiments (Gillespie, 1971; Kudo, 1976) to facilitate detection of among-treatment differences in uptake rates.

To address these questions, eleven independent microcosms were prepared and set up as follows (Table 8):

Table 8

EXPERIMENTAL DESIGN FOR EXPERIMENT 1 (YELLOW PERCH)

Treatment Number	Description	No. Replicates
1	5 kg SIL sediment, 30 L water	2
2	5 kg SIL sediment, 250 g moss, 30 L water	2
3	5 kg SIL sediment, spiked to 1 ug/g, 30 L water	2
4	control, 30 L water only	2
5	5 kg Guelph L. sediment, 30 L water	1
6	5 kg Guelph L. sediment, 250 g moss, 30 L water	1
7	5 kg Guelph L. sediment, spiked to 1 ug/g, 30 L water	1

a) Each microcosm (a 40 L glass aquarium - 48.5 cm x 25.5 cm x 28.5 cm) was soap washed and then acid washed to remove all traces of contamination from the glass.

b) To each of six microcosms were added 5 kg (wet weight) of SIL sediment. The sediments in two of these tanks were also "spiked" with 1 ug of inorganic mercury (HgCl) per gram (wet weight) of sediment, while 250 g of sphagnum moss were added to two other tanks.

c) To each of three microcosms were added 5 kg of sediments collected from Guelph Lake, a reservoir approximately 100 km west of Toronto. One of these tanks was spiked, as in b), while another received 250 g of moss.

d) No sediments were added to two remaining tanks, which were to act as controls.

e) Approximately 30 L of dechlorinated tap water was then added to each microcosm. Care was taken to minimize disturbance of the sediments during the addition of water by placement of a plastic sheet at the sediment water interface (removed carefully after all water added). Some disturbance nevertheless did occur, and the tanks were left for several days to allow the suspended matter to settle.

f) Twelve yearling yellow perch, collected from Guelph Lake and weighing approximately 10 g were then added to each microcosm. These fish were fed a freeze dried brine shrimp mixture once a day. All tanks were gently aerated to maintain high dissolved oxygen levels.

At the outset of the experiment (t = 0 days), five yellow perch were weighed and frozen for mercury analysis. As well, two SIL and two Guelph Lake sediment samples and a single moss sample were taken for mercury analysis. Samples of the 'spiked' sediments were also taken to confirm that this procedure was actually effective in elevating mercury levels in sediments. On four subsequent dates (t = 5, 10, 20, and 30 days), three fish and a single sediment sample were taken from each microcosm and preserved (frozen) for subsequent analysis. Three moss samples (one from

each microcosm containing moss) were also taken and preserved for analysis. At the conclusion of the experiment (t = 30 days), one filtered and one unfiltered water sample was taken from each microcosm and preserved in 1% nitric acid for subsequent analysis. Dissolved oxygen, pH, and water temperature were monitored throughout the experiment.

All samples preserved for mercury analysis were analyzed using an LDC atomic absorption uv photometer mercury monitor. Details of the methods used for this analysis are provided in Appendix A.

4.1.2 Experiment 2: Guppies

The experimental design for the second experiment (Table 9) was very similar to that used in the first experiment except for the following differences:

a) All sediments were spiked to at least 1 ug/g (wet weight) inorganic mercury.

b) Each experimental treatment (see Table 9) was replicated in three microcosms.

c) No "control" microcosm was used. Fish sampled at t = 0 were presumed to be "control" fish.

d) Fish were sampled at 5 intervals (t = 5, 10, 15, 20, and 30 days) rather than only four (t = 15 days added).

e) Fish were fed commercial dry flake tropical fish food.

f) Only female guppies were used to avoid differences between sexes confounding results.

g) Sediments and moss were sampled for mercury analysis only at t = 0 and t = 30 days.

Before turning to the results of these experiments, it is worth stressing another potentially important difference between the microcosm experiments and the field studies reported here (Section 5) and elsewhere

Table 9

EXPERIMENTAL DESIGN FOR EXPERIMENT 2 (GUPPIES)

Treatment Number	Description	No. Replicates
1	5 kg Guelph L. sediment	3
2	5 kg Guelph L. sediment, 100 g moss	3
3	5 kg Guelph L. sediment, 200 g moss	3
4	5 kg L. Scugog sediment	3
5	5 kg L. Scugog sediment, 100 g moss	3
6	5 kg L. Scugog sediment, 200 g moss	3

(e.g., mesocosm studies at SIL). The small size of the fish used in these experiments necessitated that the entire fish be used in the mercury analysis, rather than only muscle tissue. While methyl mercury predominates in muscle tissue (> 90%), this is not necessarily the case in other tissues. Inorganic forms of mercury may behave very differently in fish tissues than methyl mercury (i.e., shorter residence times) and thus the use of whole fish may have an important influence on the observed mercury time dynamics reported below.

4.2 RESULTS

4.2.1 Experiment 1: Perch

The most obvious outcome of the perch experiment is that, if anything, mercury levels were lower in perch exposed to sediments or moss (Table 10, Figure 6) than in controls. A sample of three perch taken at t = 0 showed higher mercury levels than in all but two samples taken subsequent to this time, for any treatment other than the control.[1] As well, the control microcosms showed consistently higher levels than were observed in the treatment tanks, again with only two exceptions.

Statistical analyses of the data support this conclusion. Since the variance in mercury levels within treatments was positively correlated with mean values, the data were subjected to a logarithmic transformation prior to statistical analysis. Analyses of variance (ANOVA's) to test for differences among mean mercury levels at different times during the experiment demonstrate significant results ($\alpha = .05$) for only treatments 6 and 7 (Table 11). Note that these analyses do not include time t = 0, since the initial sample was not associated with any particular treatment. Thus these ANOVA's do not test whether levels changed signifi-

[1] five perch were actually sampled to t = 0, but two yielded extremely high mercury levels (1.622 and 0.602 mg/kg) and were not included to the means used in Table 10 and Figure 6. Their inclusion would simply exaggerate the difference between t = 0 and other samples.

Table 10

MEANS AND STANDARD DEVIATIONS OF OBSERVED MERCURY CONCENTRATIONS IN YELLOW PERCH TISSUE SAMPLES (mg/kg) FROM EXPERIMENT 1 AT VARIOUS SAMPLING TIMES (t)

Treatment	t=0	t=5	t=10	t=20	t=30
Control	.203 ± .040	-*	-	-	-
1 SIL sediment	-	.155 ± .055	.122 ± .020	.121 ± .014	.110 ± .018
2 SIL sediment + moss	-	.136 ± .028	.138 ± .024	.126 ± .021	.147 ± .031
3 SIL sediment, spiked	-	.131 ± .030	.134 ± .036	.144 ± .020	.151 ± .056
4 Control, no sediment	-	.207 ± .050	.178 ± .024	.201 ± .023	.196 ± .020
5 Guelph sediment	-	.156 ± .046	.141 ± .010	.144 ± .034	.109 ± .014
6 Guelph sediment + moss	-	.288 ± .031	.145 ± .020	.188 ± .064	.160 ± .011
7 Guelph sediment, spiked	-	.160 ± .037	.248 ± .036	-	-

* a dash (-) indicates that fish were unavailable or not measured at this combination of time and treatment

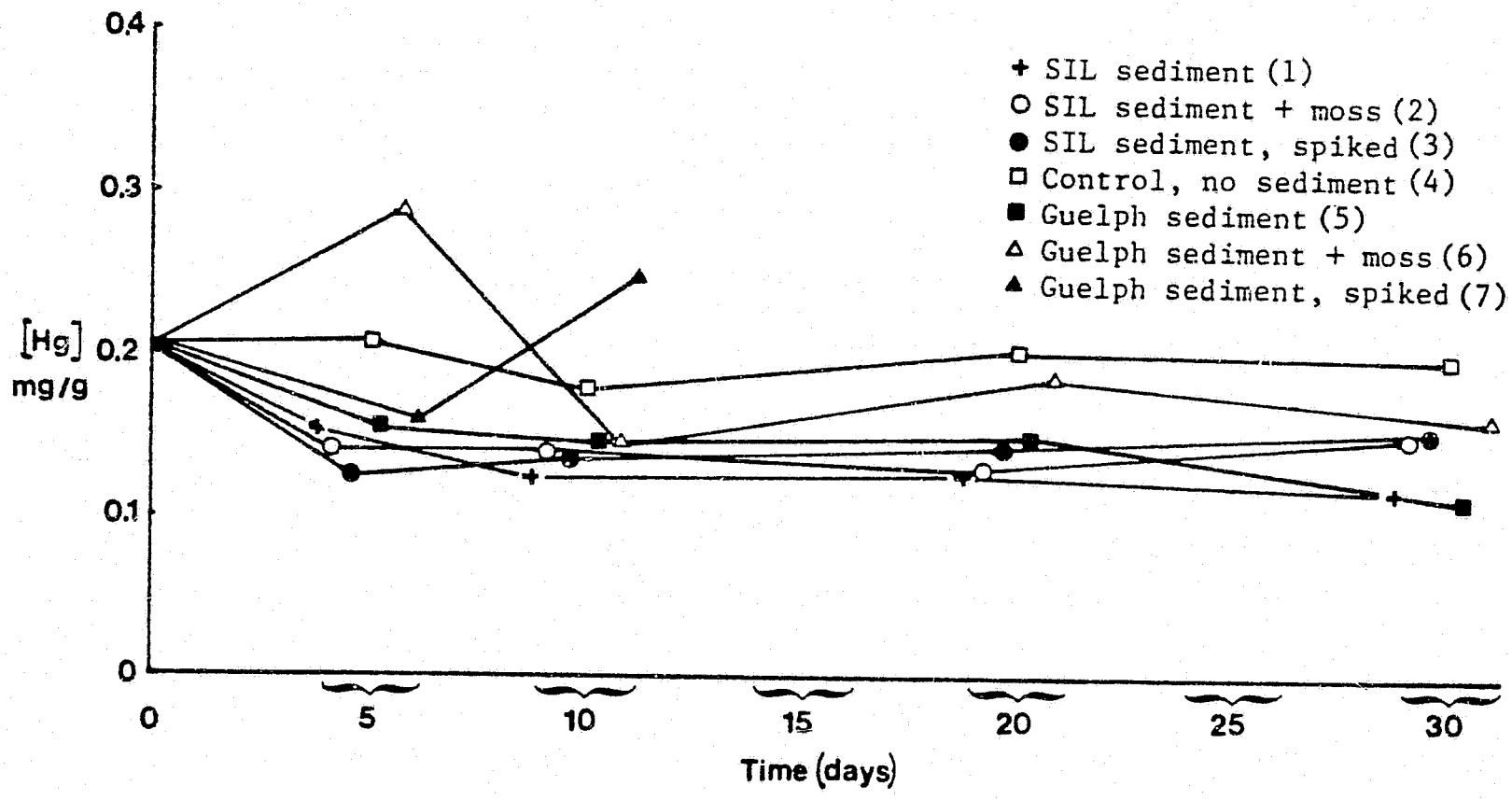


Figure 6 Mean fish mercury values as a function of time for perch used in microcosm experiment 1. Numbers in brackets on legend correspond to treatment numbers for Tables 8, 10, 11, and 12.

Table 11

RESULTS OF ANOVA COMPARING MERCURY LEVELS ($\log_{10}$ TRANSFORMED) AT DIFFERENT TIMES FOR EACH TREATMENT, FOR THE YELLOW PERCH EXPERIMENT

Treatment	DF	$F(\bar{\mu}_1=\bar{\mu}_2=\bar{\mu}_3\ldots)$	$p(F_{calc})$	$F(\sigma_1^2=\sigma_2^2\ldots)$	$p(F_{calc})$
1. SIL sediment only	3,19	2.51	.089	1.42	.236
2. SIL sediment plus moss	3,20	0.57	.639	0.20	.897
3. SIL sediment, spiked	3,19	0.60	.623	0.88	.449
4. no sediment	3,18	0.73	.547	1.48	.218
5. Guelph sediment only	3,8	1.55	.274	1.10	.350
6. Guelph sediment plus moss	3,8	8.15	.008 *	1.38	.252
7. Guelph sediment, spiked	1,4	8.44	.044 *	0.24	.627

* significant at $\alpha = .05$

cantly from initial levels, but rather whether they changed during the course of the experiment after day 5. Apparently any significant changes in treatments 1-5 (presumably declines, if any) occurred prior to day 5.

ANOVA's to test whether differences occurred among treatments produced significant results at all sampling times (Table 12). As well as the overall ANOVA's, a series of orthogonal contrasts (comparisons among means) are presented to compare individual treatment or combinations of treatments. These contrasts indicate that the control differed significantly from all treatments combined (F), for the last two sampling dates (t = 20,30), and that the Guelph treatments differed from the SIL treatments on all but the final sampling date (E). As well, there is some indication of an effect of moss additions and spiking for both sediment types, by the end of the experiment (C, D; t = 30).

4.2.2 Experiment 2: Guppies

In contrast to the perch experiment, considerable mercury uptake appears to have taken place in guppies in all treatments (Table 13, Figure 7). While this result was expected, comparisons among treatments yield unanticipated patterns. The "Guelph" treatments (sediments taken from a reservoir) tended to exhibit less uptake than the "Scugog" treatments (natural lake sediments). More surprisingly perhaps, the addition of moss to the microcosms appears to have inhibited uptake, either by slowing the rate of uptake for the "Guelph" treatments, or by lowering the maximum level for the "Scugog" treatments. These conclusions are once again supported by statistical analysis of the data which demonstrate significant differences among sampling times for most treatments (four of six, Table 14), and significant differences among treatments, at all times (Table 15). The greatest differences among treatments appear when the "Guelph" treatments are contrasted to the "Scugog" treatments (orthogonal contrast 5, Table 15). As the mean concentration for all treatments is significantly higher than the mean for the sample taken at time 0, (t_{calc} = 5.25, $p < .001$) there is little doubt that significant uptake did occur in treatments four and six, which did not exhibit

Table 12

RESULTS OF ANOVA COMPARING MERCURY LEVELS (log_{10} TRANSFORMED) AMONG TREATMENTS, AT EACH SAMPLING TIME, FOR THE YELLOW PERCH EXPERIMENT. INCLUDED WITH THE ANOVA RESULTS ARE THE OUTCOMES OF SIX ORTHOGONAL CONTRASTS.

						p(F) for orthogonal contrasts (numbers indicate treatments)					
						A	B	C	D	E	F
Time	DF	$F(\bar{y}_1=\bar{y}_2\ldots)$	$p(F_{calc})$	$F(\sigma_1^2=\sigma_2^2\ldots)$	$p(F_{calc})$	2 vs 3	6 vs 7	1 vs 2+3	5 vs 6+7	1+2+3 vs 5+6+7	4 vs 1+2+3+5+6+7
5 days	6,26	5.21	.001 *	0.37	.901	.769	.006 *	.301	.061	.003 *	.053
10 days	6,26	6.38	<.001 *	0.96	.453	.664	.002 *	.332	.039 *	.001 *	.052
20 days	5,24	7.49	<.001 *	0.97	.431	.187	--	.254	.088	.013 *	<.001 *
30 days	5,20	7.25	<.001 *	0.79	.557	.743	--	.004 *	.011 *	.911	<.001 *

Treatments:

1 - SIL sediment
2 - SIL sediment plus moss
3 - SIL sediment plus 1 ug/g Hg
4 - control, no sediment
5 - Guelph sediment
6 - Guelph sediment plus moss
7 - Guelph sediment plus 1 ug/g Hg

* significant at $\alpha = .05$

Table 13

MEANS AND STANDARD DEVIATIONS OF OBSERVED MERCURY CONCENTRATIONS IN GUPPIES (mg/kg) FROM EXPERIMENT 2 AT VARIOUS SAMPLING TIMES (t)

Treatment	t=0	t=5	t=10	t=15	t=20	t=30
Control	.038 ± .020	-*	-	-	-	-
1 Guelph, no moss	-	.149 ± .031	.150 ± .032	.221 ± .014	.111 ± .038	.129 ± .015
2 Guelph, low moss	-	.163 ± .073	.155 ± .066	.236 ± .111	.118 ± .055	.131 ± .018
3 Guelph, high moss	-	.047 ± .026	.087 ± .012	.239 ± .131	.082 ± .013	.112 ± .032
4 Scugog, no moss	-	.311 ± .304	.404 ± .199	.598 ± .396	.290 ± .167	.401 ± .153
5 Scugog, low moss	-	.086 ± .039	.375 ± .220	.337 ± .188	.218 ± .067	.164
6 Scugog, high moss	-	.139 ± .028	.210 ± .090	.216 ± .054	.182 ± .107	.163 ± .017

* a dash (-) indicates that fish were unavailable or not measured at this combination of time and treatment

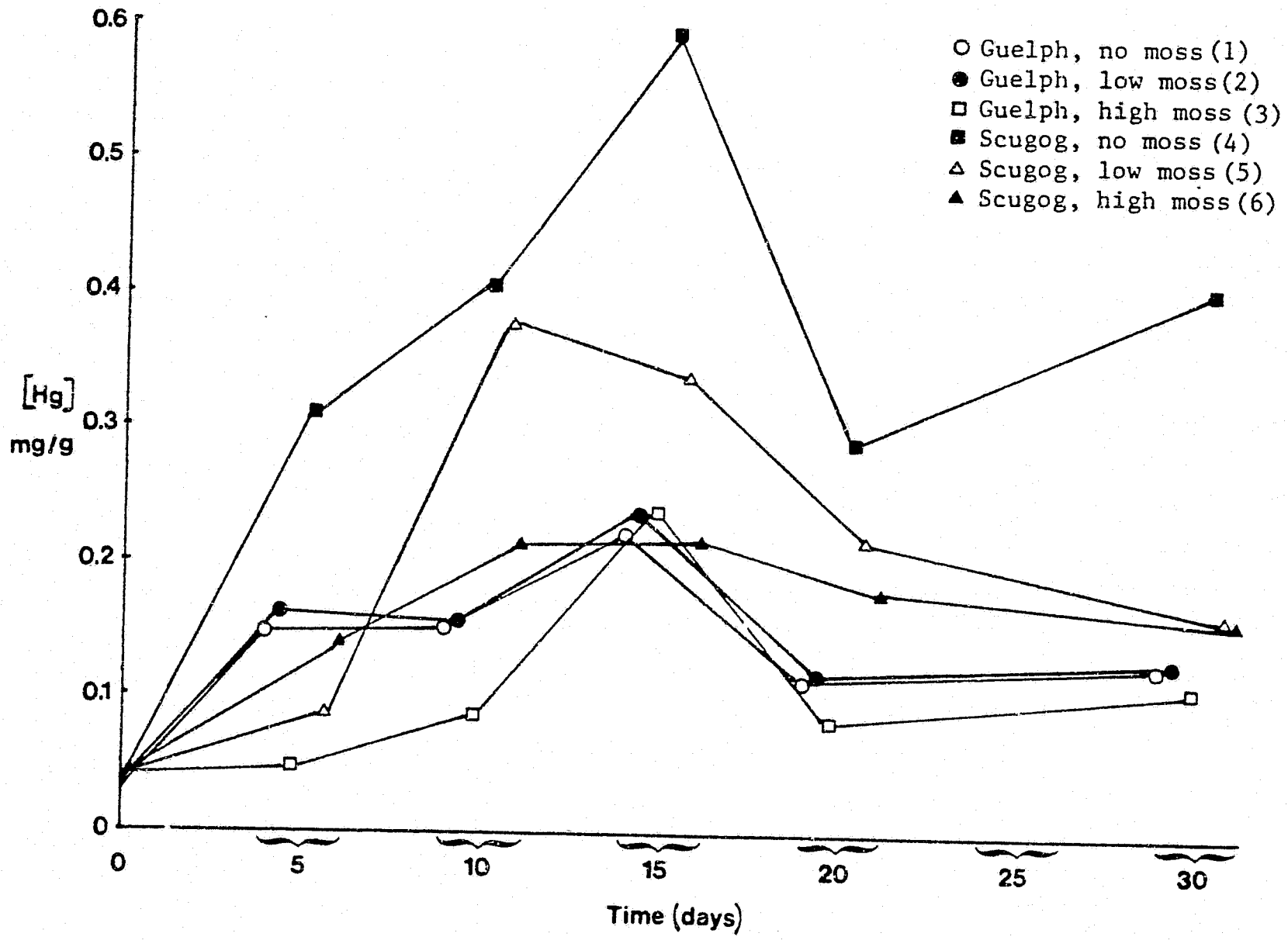


Figure 7 Mean fish mercury values as a function of time for guppies used in microcosm experiment 2. Numbers in brackets correspond to treatment numbers for Tables 9, 13, 14, and 15.

Table 14

RESULTS OF ANOVA COMPARING MERCURY LEVELS ($\log_{10}$ TRANSFORMED) AT DIFFERENT TIMES, FOR EACH TREATMENT, FOR THE GUPPY EXPERIMENT

Treatment	DF	$F(\bar{\mu}_1=\bar{\mu}_2...)$	$p(F_{calc})$	$F(\sigma_1^2=\sigma_2^2...)$	$p(F_{calc})$
1. Guelph sediment	4,39	6.04	<.001 *	2.09	.079
2. Guelph sediment plus low moss	4,40	3.75	.011 *	2.04	.086
3. Guelph sediment plus high moss	4,38	4.75	.003 *	3.03	.017 *
4. Scugog sediment	4,31	2.31	.080	0.50	.737
5. Scugog sediment plus low moss	4,32	8.59	<.001 *	1.63	.180
6. Scugog sediment plus high moss	4,40	2.34	.071	5.22	<.001 *

* significant at α = .05

Table 15

RESULTS OF ANOVA COMPARING MERCURY LEVELS ($\log_{10}$ TRANSFORMED) AMONG TREATMENTS, AT EACH SAMPLING TIME, FOR THE GUPPY EXPERIMENT. INCLUDED WITH THE ANOVA RESULTS ARE THE OUTCOMES OF FIVE ORTHOGONAL CONTRASTS.

						p(F) for orthogonal comparisons (numbers indicate treatments)				
Time	DF	$F(\bar{\mu}_1=\bar{\mu}_2\ldots)$	$p(F_{calc})$	$F(\sigma_1^2=\sigma_2^2\ldots)$	$p(F_{calc})$	A 2 vs 3	B 5 vs 6	C 1 vs 2+3	D 4 vs 5+6	E 1+2+3 vs 4+5+6
5 days	5,48	9.11	<.001 *	7.22	<.001 *	<.001 *	.098	.012 *	.007 *	.027 *
10 days	5,48	12.41	<.001 *	0.94	.451	.009 *	.045 *	.145	.061	<.001 *
15 days	5,48	4.34	.003 *	2.86	.014 *	.504	.246	.586	.002 *	.004 *
20 days	5,45	7.37	<.001 *	0.63	.677	.207	.255	.383	.134	<.001 *
30 days	5,31	9.12	<.001 *	6.58	<.001 *	.069	.968	.333	.002 *	<.001 *

Treatments:

1 - Guelph sediment
2 - Guelph sediment plus low moss
3 - Guelph sediment plus high moss
4 - Scugog sediment
5 - Scugog sediment plus low moss
6 - Scugog sediment plus high moss

* significant at $\alpha = .05$

significant changes through time subsequent to day 5 (Table 14).

4.3 DISCUSSION

The perch experiment provides some, albeit weak, support for the prediction that additions of moss would stimulate mercury uptake by biota (Table 12, contrasts C, D; t = 30). This result is almost completely overwhelmed, however, by the overall reductions in perch mercury levels that appear to have occurred for all treatments (relative to the control), especially those using SIL sediments. An explanation for this result may derive from the nature of the sediments used in this experiment, and from the behaviour of the experimental fish. The SIL sediments contained a high proportion of extremely fine particles, some of which remained in suspension throughout the experiments. As well as simply exhibiting very low sedimentation rates, periodic disturbance of the bottom by perch resulted in ongoing resuspension of these particles. This behaviour also took place in the "Guelph" treatments, resulting in higher suspended sediment levels in these treatments than might be expected naturally. The problem was less pronounced for the "Guelph" treatments, however, due to the lower proportion of fine material in these sediments.

The presence of suspended inorganic matter in the water column has been shown to reduce the availability of mercury to biota (Rudd *et al.*, 1983), presumably by binding otherwise bioavailable mercury to inorganic particles. In the perch experiment, the treatments subjected to the highest levels of suspended sediments (unfortunately, we have no quantitative measure of suspended sediment concentrations) appear to show the most obvious declines in fish mercury levels. Apparently, the mercury bioavailability in the SIL microcosms was less than the levels to which the experimental fish were exposed in their natural habitat (Guelph Lake), thus resulting in a net loss of mercury from these fish. While this is an *ad hoc* explanation, it is further supported by the fact that the effect (declines in fish mercury) was less pronounced in the "Guelph" treatments where suspended sediment levels appeared to be much lower. Since the experiments were not explicitly designed to test this

hypothesis (i.e. that suspended sediments inhibit mercury uptake by biota), it is not possible to draw any firm conclusions. Recent results from mesocosm experiments at SIL, however, suggest that suspended inorganic matter actually enhances uptake (R.E. Hecky, personal communication), in contrast to the observations of Rudd et al. (1983).

Changes in the choices of both assay species and sediment types for the second experiment effectively removed this "suspended sediment effect" from the experimental results. In this experiment, uptake of mercury was observed for all treatments, and significant differences were found both between sediment types and among the treatments with varying amounts of moss added. It is likely that the change of species was of greatest significance, since Guelph Lake sediments were used in both experiments. Guppies, unlike perch, were not observed to disturb the sediment.

Additions of moss appear to have reduced the rate of uptake (Guelph sediments) or lowered the maximum mercury level (Scugog sediments). This result was contrary to expectation, our research hypothesis being that additions of organic matter would stimulate microbial activity and thus methylation rates, thereby increasing the availability of mercury for uptake by biota. An alternative hypothesis, which would explain the observed results, is that methylation rates are adequate to cause uptake without moss additions and that the moss actually acts as an alternate sink for sediment-derived methyl mercury, thereby reducing the supply of mercury to the water column where fish reside. In support of this hypothesis, we observed substantial increases in the concentrations of mercury in moss between the start and the end of the experiment (Table 16).

The differences among sediment types in uptake by guppies are more difficult to explain. The organic content of the Scugog sediments (measured as % loss on ignition) is actually lower (0.47-0.51%) than observed for the Guelph sediments (8.6-9.4%). One would expect organic matter in the sediments to stimulate methylation, thus leading to higher uptake rates for sediments with higher organic content. Measured mercury levels in

Table 16

MERCURY CONCENTRATIONS IN MOSS AT THE START OF EXPERIMENT 2 (CONTROL), AND AT THE END OF EXPERIMENT 2 FOR LOW AND HIGH MOSS TREATMENTS OF TWO SEDIMENT TYPES

Treatment		Tank	Time	[Hg] mg/kg
	Control	0	Start	0.0050
2	Guelph, low moss	4	End	0.1893
2	Guelph, low moss	5	End	0.0664
2	Guelph, low moss	6	End	0.1444
3	Guelph, high moss	7	End	0.1059
3	Guelph, high moss	8	End	0.0819
3	Guelph, high moss	9	End	0.1849
5	Scugog, low moss	13	End	0.2072
5	Scugog, low moss	14	End	0.5247
5	Scugog, low moss	15	End	0.2409
6	Scugog, high moss	16	End	0.1244
6	Scugog, high moss	17	End	0.1401
6	Scugog, hgih moss	18	End	0.1108

the sediments at time t = 0 were lower in the Scugog sediments (.04-.06 ug/g) than the Guelph sediments (.08-.10 ug/g). Other water quality parameters (pH, D.O., temperature) were not significantly different among treatments. Presumably some other unmeasured characteristic of the Scugog sediments, such as particle size or the composition of the microflora, is responsible for the observed differences among sediment types.

In conclusion, the two microcosm experiments both produced results which were contrary to our a priori expectations. These results indicate that the use of microcosms as a tool for routine assays of the effects of individual factors such as soil and vegetation type is far from straightforward. On the other hand, both experiments yielded informative results. The perch experiment is consistent with the observations of Rudd et al. (1983) that suspended inorganic matter can inhibit mercury uptake by fish but differs from more recent mesocosm results. Additions of suspended matter have been suggested (Rudd et al., 1983) as a method of mitigation of the mercury problem. The guppy experiment demonstrated that organic matter may actually act as an alternate sink for mercury derived from sediments, thereby reducing the supply of mercury to other biota. This phenomenon is almost certainly a transitory one, however, since the organic matter will eventually decompose and thus release the mercury stored within its tissues to other ecosystem compartments.

Unfortunately, the results of the microcosm experiments do little to argue strongly in favor of their use as a routine assay tool. Although both sets of experiments produced significant treatment effects, the results were often counter-intuitive (at least initially), and certainly differed from results obtained in similar mesocosm experiments. At the second workshop, it was suggested that the expected treatment effects might have been observed more clearly, if the experiments had lasted longer. At the time of their design, however, a thirty day duration was considered adequate. Certainly any future microcosm experiments should be carried out over longer periods of time.

At this point in time, then, it is clear that the use of microcosms for

assays to examine factors influencing mercury uptake in newly created reservoirs is very much in the developmental stage. Further refinements of the methodology are necessary to avoid the influence of confounding factors such as resuspension of fine sediments. Of the two experiments conducted in this study, the second produced more useful results, and shows more promise as an assay method. Nevertheless, even the results of this experiment strongly suggest that careful attention be given to the detailed design of the assay (i.e. How are the sediments being tested different? Should fresh, uncontaminated vegetation be used?).

5 FIELD DATA COLLECTION

5.1 METHODS

Eleven reservoirs were selected from five provinces as sites for field data collection during the summer of 1984 (Table 17). A map illustrating their locations is provided in Section 6 (Figure 8). Throughout Canada, construction of the earliest hydroelectric projects typically took place in locations relatively near population centres. More recently, utilities have had to choose more remote locations as development sites. Because of this, many desirable locations (i.e. easily accessible) were excluded from consideration due to age or proximity to possible sources of mercury contamination. Between July and September of 1984, samples of water, sediment, and fish flesh were taken from the eleven reservoirs.

5.1.1 Water

Mercury levels in water uncontaminated from local sources tend to be low (normally below detection limits), even in reservoirs with elevated fish mercury concentrations. Nevertheless, we considered it worthwhile measuring mercury levels in the water at field sites to eliminate the possibility of local contamination.

In reservoirs which were thermally stratified (Gisborne, Long, Aubrey, and Lower Arrow Lakes), water samples were collected at one location from the hypolimnion, metalimnion, and epilimnion, then mixed. In unstratified lakes (all other seven reservoirs), samples were taken at one location of water from 0.5 m below the surface, midwater, and near-bottom locations, and mixed to provide a composite sample. The appropriate process was repeated at two other sites to yield three water samples from each reservoir. Each of the three 150 ml water samples collected was preserved with sulphuric acid and dichromate solution and stored in glass containers with teflon-lined caps. Samples were refrigerated prior to and during transport to the University of Toronto, where laboratory analysis took place. Details of the laboratory techniques used are given

Table 17

NAMES AND LOCATIONS OF RESERVOIRS SAMPLED IN 1984*

Reservoir Name	Province	Latitude	Longitude
Cheticamp	Nova Scotia	4635	6030
Gisborne	Nova Scotia	4635	6030
McMillan	Nova Scotia	4635	6030
Wreck Cove	Nova Scotia	4635	6030
Long	New Brunswick	4702	6654
Aubrey	Ontario	4655	8313
Tunnel	Ontario	4627	8326
Red Rock	Ontario	4619	8317
Brazeau	Alberta	5255	11540
Lower Arrow	British Columbia	4930	11810
Libby	British Columbia	4900	11510

* Post-impoundment physical and chemical data for the reservoirs is provided in Appendix C.

in Appendix A.

5.1.2 Sediments

Sediment samples for each reservoir consisted of composites of no less than three Eckman dredge collections taken from the middle of each of three major sections or basins of each reservoir. Maps showing sampling locations for both sediments and fish are provided in Appendix D. Sample depth varied between reservoirs and basins, and ranged from 6 to 40 m. In instances where the substrate was hard, gravelly, or strewn with wood debris, as many as twelve Eckman grabs were combined to yield a sample sufficient for analysis. At each location, the samples were mixed by hand in a plastic tub, and a subsample was selected and stored in an individually labelled plastic bag and refrigerated prior to and during transport to the University of Toronto. Laboratory analytical methods are described in Appendix A.

5.1.3 Fish

Table 18 provides information on the fishing effort, gear used, and depths of sampling for the fish collections. The number of gill nets set (3-23), total hours fished (12.5-240.0), mesh size, and depths of gill nets set (0-15 m) varied considerably among reservoirs, and reflected the species being sought and the ease with which they were obtained. Only those numbers and species of fish required for sampling were retained. The following information was recorded for each fish retained: length (mm), weight (g), sex, maturity, and stomach contents. Scale samples were also taken from each of these fish; however, because of time and budgetary restrictions, fish were not aged. A 30 g skinless muscle fillet was taken from each fish retained for mercury analysis. The fish flesh samples were stored in individually labelled whirlpack bags and frozen for later analysis. Methods used to analyze mercury in fish flesh were identical to those used for the microcosm fish, as described in Appendix A.

Table 18

FISHING EFFORT, GEAR USED, AND DEPTHS
OF SETS FOR FISH COLLECTIONS DURING 1984

Reservoir	# Sets	Total # Hours Fished	Type of Gillnet Used*	Depths of Sets
Cheticamp	3	12.5	C	2- 5 m
Gisborne	2	20.0	C	2- 5 m
McMillan	2	28.0	C	2- 4 m
Wreck Cove	5	46.0	C	2- 5 m
Long	6	84.0	A,C	0- 5 m
Aubrey	20	214.0	A,B,C	1-15 m
Tunnel	22	240.0	A,B,C	1-15 m
Red Rock	23	185.0	A,B,C,D	1-15 m
Brazeau	4	60.0	A,B,C,E	2-10 m
Lower Arrow	15	174.0	A,B,C,E	0-10 m
Libby	6	91.0	A,B,C,E	0-10 m

* A - 7.5 m x 2 m panels of 51, 64, 76, 89 mm stretched monofilament mesh
B - 15.0 m x 2 m panels of 51, 76, 102, 127 mm stretched monofilament mesh
C - 15.0 m x 2 m panels of 76, 102, 127 mm stretched monofilament mesh
D - 15.0 m x 10 m panel of 114 mm stretched monofilament mesh
E - 100.0 m x 2 m panel of 89 mm stretched multifilament twine mesh

Attempts were made during the field collections to obtain 25 individuals of the following taxonomic groups from each reservoir: salmonids, another top carnivore, coregonids, and catastomids. Our intention in collecting fish from more than one trophic level was to examine the hypothesis that biomagnification of mercury occurs in the aquatic food chain (Wren et al., 1983) (i.e. that mercury levels in reservoir fish increase with trophic level). As well, we felt that data for more than one species might increase the opportunity for inter-reservoir comparisons, since not all reservoirs contain the same species. In some cases, however, it was only possible to obtain samples from one or two of the taxonomic groups listed above. Wherever possible, at least 10 individuals from each of these groups were analyzed for mercury.

5.2 RESULTS

Table 19 provides the means and standard deviations of the lengths, weights, and mercury concentrations for the fish species of primary interest, subsampled from those taken during the field collections of 1984. Results of the laboratory analyses on sediment samples are presented in Table 20. Most of the interpretation of these results is left to Section 7; some of the highlights, however, are briefly discussed below.

The four reservoirs in the Wreck Cove development are of the same order of magnitude in size (355-1050 ha) and were impounded in the same year (1978), yet one reservoir of the four (Wreck Cove Lakes Reservoir) yielded brook trout with mean mercury concentrations (0.94 ± 0.38 mg/kg) considerably above those in the other three reservoirs (0.36 ± 0.13, 0.23 ± 0.04, and 0.19 ± 0.07 mg/kg in Cheticamp, Gisborne, and McMillan Reservoirs, respectively). As discussed in Section 7, this result could not be traced to any obvious physical or chemical differences among these reservoirs; further monitoring is planned to explore this question (T. Toner, personal communication). Other mean mercury values greater than 0.5 mg/kg were only observed in samples of piscivorous fish dominated by larger individuals (walleye in Red Rock and Tunnel Lakes; northern pike

Table 19

MEANS AND STANDARD DEVIATIONS OF THE LENGTHS, WEIGHTS AND MERCURY CONCENTRATIONS OF FISH COLLECTED DURING THE 1984 FIELD WORK AND SELECTED FOR MERCURY ANALYSIS

Reservoir	Species	Number Fish Caught	Number Fish Analyzed	Mean Values ± S.D. for Analyzed Fish Length (cm)	Weight (kg)	[Hg] (mg/kg)
Cheticamp	Brook Trout	24	10	22.63 ± 1.30	0.12 ± 0.01	0.36 ± 0.13
Gisborne	Brook Trout	20	10	22.10 ± 1.66	0.13 ± 0.03	0.23 ± 0.04
McMillan	Brook Trout	20	10	19.88 ± 1.90	0.09 ± 0.03	0.19 ± 0.07
Wreck Cove	Brook Trout	21	10	22.24 ± 1.19	0.13 ± 0.02	0.94 ± 0.38
Long	Brook Trout	4	4	22.95 ± 2.97	0.17 ± 0.07	0.19 ± 0.13
	Lake Trout	16	10	47.67 ± 5.97	1.47 ± 0.75	0.42 ± 0.17
	White Sucker	21	10	25.10 ± 4.40	0.22 ± 0.10	0.23 ± 0.09
Aubrey	Northern Pike	11	11	43.18 ± 7.87	0.64 ± 0.40	0.37 ± 0.14
	Walleye	20	19	32.18 ± 4.32	0.36 ± 0.15	0.44 ± 0.10
	White Sucker	19	10	43.53 ± 2.09	1.22 ± 0.17	0.30 ± 0.08
Tunnel	Northern Pike	14	12	57.47 ± 12.11	1.64 ± 1.03	0.43 ± 0.20
	Walleye	22	21	38.68 ± 7.11	0.66 ± 0.32	0.54 ± 0.21
	White Sucker	23	10	43.67 ± 6.17	1.35 ± 0.49	0.17 ± 0.05
Red Rock	Northern Pike	8	8	57.56 ± 9.59	1.41 ± 0.64	0.73 ± 0.46
	Walleye	1	1	50.80 --	1.26 --	1.01 --
	White Sucker	19	10	43.86 ± 6.00	1.13 ± 0.27	0.23 ± 0.06
Brazeau	Northern Pike	17	17	47.27 ± 7.67	0.83 ± 0.43	0.31 ± 0.04
	White Sucker	20	10	46.00 ± 6.63	1.47 ± 0.51	0.34 ± 0.14
Lower Arrow	Mountain Whitefish	21	5	32.78 ± 1.68	0.33 ± 0.02	0.07 ± 0.01
	Longnose Sucker	7	7	33.60 ± 5.41	0.47 ± 0.18	0.22 ± 0.10
	Largescale Sucker	14	10	31.21 ± 9.74	0.45 ± 0.36	0.17 ± 0.03
Libby	Mountain Whitefish	3	3	29.40 ± 5.20	0.35 ± 0.19	0.23 ± 0.02
	Largescale Sucker	20	10	39.52 ± 1.99	0.68 ± 0.12	0.22 ± 0.05

Table 20

MERCURY CONCENTRATION AND ORGANIC CONTENT (MEASURED AS % LOSS ON IGNITION) IN SEDIMENTS FROM RESERVOIRS SAMPLED IN 1984

Reservoir	Sediment Sample #	Hg Concentration (mg/kg)	% Loss on Ignition
Cheticamp	1	0.1999	31.23
	2	0.1305	55.61
	3	0.1073	66.43
Gisborne	1	0.0987	75.41
	2	0.1311	31.61
	3	0.1023	6.36
McMillan	1	0.1258	26.20
	2	0.0950	10.69
	3	0.1089	38.73
Wreck Cove	1	0.1394	20.02
	2	0.1135	45.52
	3	0.1062	20.82
Long	1	0.1245	9.03
	2	0.2199	23.39
	3	0.1811	19.04
Aubrey	1	0.1466	20.49
	2	0.1304	13.35
	3	0.1109	17.84
Tunnel	1	0.1392	47.73
	2	0.1234	11.78
	3	0.1051	13.73
Red Rock	1	0.1002	3.01
	2	0.1016	2.55
	3	0.1183	3.83
Brazeau	1	0.1482	6.43
	2	0.1151	5.13
	3	0.1472	5.99
Lower Arrow	1	0.0808	5.83
	2	0.0445	0.62
	3	0.0960	10.34
Libby	1	0.0750	2.36
	2	0.1146	2.82
	3	0.0929	7.95

in Red Rock Lake). Mercury levels in piscivorous species generally exceeded those in bottom feeders. Similarly, the range of computed means and standard deviations for brook trout, northern pike, and walleye was larger than for whitefishes and suckers.

Meister et al. (1979) concluded that elevated mercury concentrations in fish flesh were an inevitable consequence of flooding; Abernathy and Cumbie (1977) suggested that these elevated levels would decline after 3-5 years post-impoundment. The year of impoundment for the reservoirs sampled in 1984 varied from 1952 (Long Lake) to 1978 (the four Nova Scotia reservoirs). This temporal variation, in addition to the physical and chemical descriptors identified by previous authors as related to fish flesh mercury (e.g. water hardness, Rodgers, 1982; pH, Jernelov et al., 1975; vegetation inundation, Boucher and Schetagne, 1983; area flooded, Bodaly et al., 1984b) were analyzed for each reservoir in an attempt to identify key factors contributing to mercury bioaccumulation (see Section 7).

Estimates in the literature of what constitutes natural sediment mercury values have varied widely, and may largely reflect geological variations among the areas being sampled. Stokes et al. (1983) found sediment mercury levels in Ontario lakes to range from .030 to .150 mg/kg; Thomas (1972) estimated the background mercury level of Lake Ontario basin sediments to be .300 mg/kg. In reviewing the observations of various authors reporting on mercury sediment levels from a variety of locations, Wren (1983) reported natural sediment mercury levels to range from .010 to .440 mg/kg. All of the sediments sampled during the 1984 field work had mercury levels well within this range (see Table 20). The highest value came from Long Lake (.2199 mg/kg); mercury values were lower in the two British Columbia reservoirs sampled (Lower Arrow, Libby) than elsewhere. These results do not imply the presence of unusually high mercury concentrations in the sediments of the reservoirs sampled in 1984.

No correlation was found between % loss on ignition (LOI) and mercury values in the sediments ($r < 0.1$). In some cases, there was a visible

difference in the organic content of the sediment samples taken from different parts of the same reservoir, which was mirrored in the intra-reservoir range of LOI values. For example, the highest LOI values from Gisborne and Lower Arrow reservoirs were more than 10 times the lowest values from the same reservoirs. LOI readings from sediments collected west of Ontario tended to be lower than those from the rest of the reservoirs.

Results of the water mercury analysis were much higher than expected, suggesting problems with sample contamination or analysis. These results are an illustration of the problems in analytical variability described by Richkus (1984). We have no good explanation for the high values observed, nor any reason to suspect a serious contamination problem in the reservoirs, since mercury levels observed in fish from the same reservoirs are not consistently high. Since the water mercury values obtained appear to be anomalous, they have not been used in any of the analyses discussed later in this report.

6 EXISTING DATA COLLECTION

During the first workshop held in April of 1984, several priorities were identified for work to be undertaken during the course of the project, including an emphasis on the collection and utilization of existing relevant data. As a result of these discussions, we decided to develop a database such that information gathered from existing sources, and that from the field work could be merged to maximize the opportunities for inter-reservoir comparisons.

6.1 METHODS

For all of the factors identified during the first workshop as possible determinants of mercury levels in fish flesh (Table 21), one or more parameters were suggested as means of testing these hypotheses of effect. This information was then sought out for reservoirs created in the last three decades which also had information on fish mercury levels. Of the hundreds of hydroelectric projects in Canada, twenty sites (8 in Quebec, 5 in Newfoundland, 4 in Manitoba, and 1 in each of Ontario, Saskatchewan, and British Columbia) were identified as meeting these criteria. The specific procedures used to arrive at this set of twenty sites are detailed in the next three paragraphs.

Names and years of impoundment for all hydroelectric reservoirs were solicited from each major utility. This information, in addition to that provided in the Register of Dams in Canada (Canadian National Committee of the International Commission on Large Dams, 1976), and from a series of maps identifying principal power generating developments (Energy, Mines, and Resources Canada, 1981), was used to produce a list of all sites to be considered for inclusion in the database.

The availability of data on mercury in fish from these locations was then established by contacting employees of the federal and provincial governments, universities, and utilities in each province. Since the phenomenon of mercury bioaccumulation in reservoir fish is a relatively

Table 21

FACTORS SUGGESTED AS DETERMINANTS OF MERCURY CONCENTRATIONS IN RESERVOIR FISH (FROM FIGURE 5)

Number	Factor
1	Soil type in flooded area
2	Vegetation type and amount in flooded area
3	Erosion rates
4	Relative area flooded
5	Resuspension and sedimentation
6	Other mercury sources
7	Ca, pH
8	Trophic status
9	Diet
10	Se
11	Temperature
12	Time since impoundment
13	Watershed size
14	Runoff

recent discovery, and has been suggested to be of a transient nature (Abernathy and Cumbie, 1977), only a very small proportion of the reservoirs had been sampled for fish mercury concentrations.

For recently created (since 1954) reservoirs in which fish mercury levels had been measured, data on the factors identified in Table 21 were then sought from the same sources identified above (over 100 individuals were contacted by phone or mail during this data search). Published literature, and soil and vegetation maps were used to supplement this information.

For some of these factors (Ca, pH, Se, time since impoundment, watershed size), the exact type of data to be gathered was obvious. For others the procedure was much less straightforward. To provide a basis for comparison between reservoirs from vastly different regions of vegetation and geological history, a simple classification scheme was developed to describe the soils and vegetation in the flooded zone. Flooded soils were categorized by % flooded area in each of five categories: rock or overburden; sand and gravel; clay; silt; and other. Four categories of vegetation were used to describe the flooded zone, % area in each of: coniferous stands; deciduous stands; open (but vegetated) land; and area cleared of vegetation. Fetch was used as a measure of erosion (assuming greater erosion would be associated with greater fetch). The relative area flooded was reflected by a within-reservoir comparison of pre- and post-impoundment surface area, volume, mean depth, and maximum depth. In addition, data on the maximum, minimum, and average drawdowns provided a picture of the post-impoundment water level fluctuations. Pre- and post-impoundment turbidity, secchi depth, total dissolved solids (TDS) and total suspended solids (TSS) were used as indicators of changes in resuspension and sedimentation. Reservoirs which were subject to inputs of anthropogenically-derived mercury were excluded from consideration. Trophic status indicators used were pre- and post-impoundment total phosphorous and chlorophyll a. The presence/absence of representatives of six major fish taxonomic groups (salmonids, whitefishes, esocids, suckers, centrarchids, and percids), collectively called fish community

present, was the only indicator of diet (factor 9), however indirect, available on an extensive basis. Extreme variations in the time of year and locations selected for on-site measurements of temperature required that a more generalized method of comparing temperature regimes be selected: in this case, we used a 30 year average of annual and growing season degree days from the closest Canadian Climate Normals Station (Environment Canada, 1982). For similar reasons, runoff values were taken from maps developed by Fisheries and Environment Canada (1978). Sediment mercury concentrations were available for 4 of the 20 reservoirs in the existing data set (Smallwood, Ossokmanuan, Hinds Lake, and SIL).

Equivalent physical, chemical, and biological information was collected either through existing data sources or during 1984 for the 11 reservoirs included in the field work.

6.2 RESULTS

In all, thirty-one reservoirs from all temperate regions of Canada were included in the database (Figure 8). A set of three files for each reservoir was used to accommodate all of the data collected from existing sources and from the field work of 1984. The first file held all of the physical and chemical data (with the exception of sediment mercury values) as described in Section 6.1. The second file contained sediment mercury data (this file was used only for the 11 field-sampled sites, and the four additional reservoirs for which existing sediment mercury data were available). The third file incorporated information on the species, sex, length, weight, age, and mercury concentration of each fish sampled for mercury analysis. When all existing and field data had been placed in the appropriate files and checked for data entry errors, all three file types for all reservoirs were merged into a master file to facilitate the statistical analyses described in Section 7.

The availability of each type of data varied from 3% (for pre-impoundment selenium) to 100% for a number of factors. For those pieces of information taken from Canada-wide data summaries or maps (number of annual and

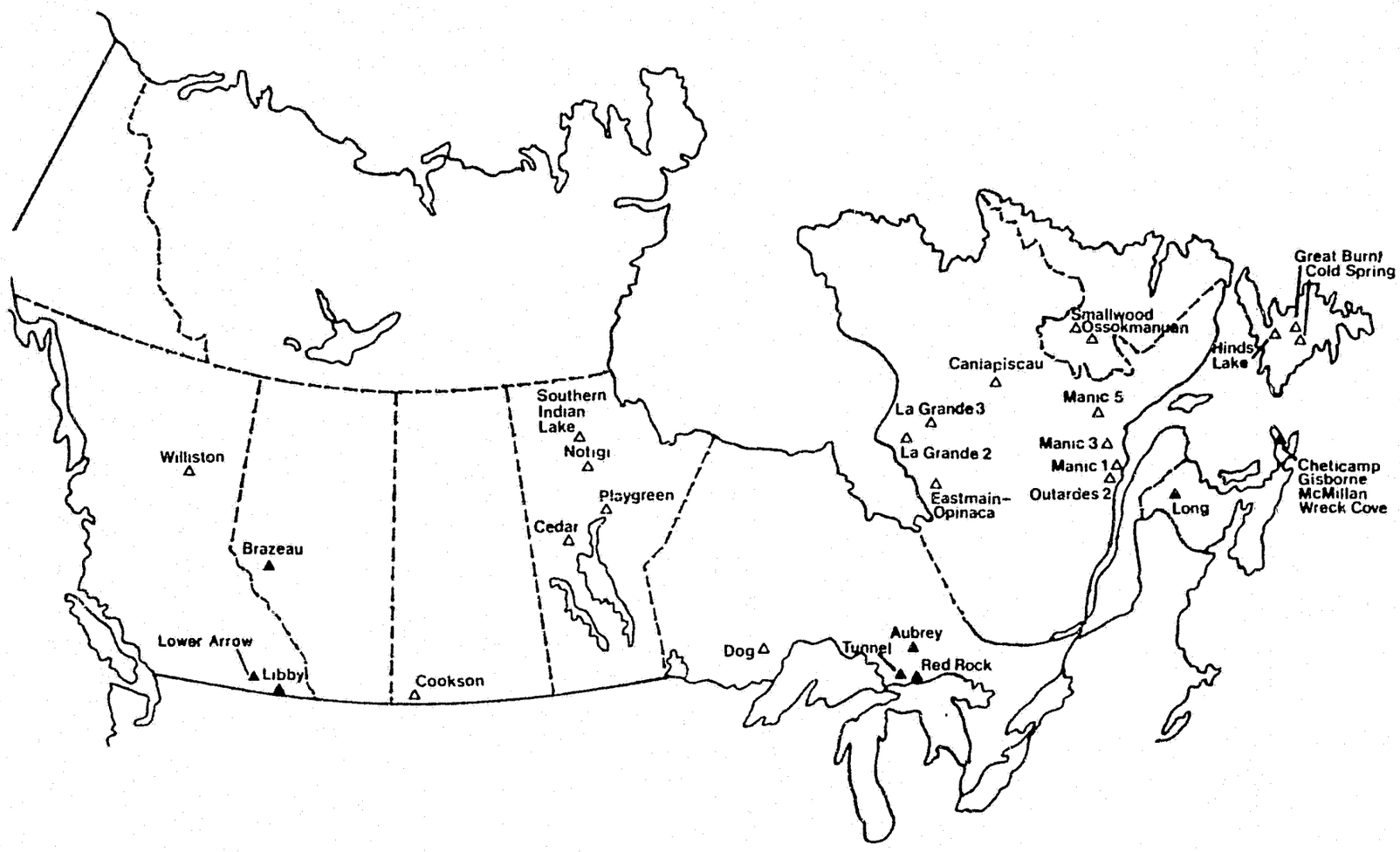


Figure 8 Location of the reservoirs used in the database. △ denotes reservoirs for which existing data was available. ▲ indicates the reservoirs sampled in 1984.

growing season days, runoff), data could be extracted for each reservoir. Similarly, some values were universally available at reservoirs (maximum drawdown, fish community present, time of impoundment). There was generally more data available from the post-impoundment period relative to pre-impoundment. Some regions had greater data availability than others (e.g., the only Alberta reservoir (Brazeau) had little water chemistry information available). Lack of data obviously hinders the opportunity to test certain hypotheses over a broad range of reservoirs; this proved to be especially true of some of the water chemistry parameters. Less than 30% of the reservoirs had the following information available: pre- and post-impoundment turbidity, pre-impoundment secchi depth, pre-impoundment TSS, pre-impoundment chlorophyll a, pre- and post-impoundment selenium. Table 22 illustrates the number of statistical among-reservoir comparisons possible with the compiled data.

Table 23 provides information on the number of years in which fish were sampled and total number of samples of each of the major fish species included in the master database; the number of sediment samples is also specified. The greatest opportunity for among-reservoir comparisons of fish mercury was for pike (19 of 31 reservoirs); other species for which mercury values had been measured in more than 10 reservoirs were lake whitefish (15), walleye (13), and white sucker (11). Only two reservoirs (Smallwood and Ossokmanuan) had fish mercury data for more than 4 of the species listed in Table 23; eight reservoirs (Hinds Lake, Cold Spring, Great Burnt, Cheticamp, Gisborne, McMillan, Wreck Cove, Dog) hadddata for only 1 species. Within reservoirs, only one year of data was available for any given species in over 65% of the cases. Five or more years of data were available for only 7% of the combinations of reservoir and species; this level of multi-year data availability only occurred in Manitoba reservoirs.

Table 22

AVAILABILITY OF PHYSICAL, CHEMICAL, AND BIOLOGICAL DATA FOR THE TWENTY RESERVOIRS IN THE EXISTING DATA SET PLUS THE ELEVEN RESERVOIRS SAMPLED IN 1984

Reservoir	% soil type	% veg type	fetch	pre surface area	post surface area	pre volume	post volume	pre mean depth	post mean depth	pre max depth	post max depth	max drawdown	min drawdown	avg drawdown	pre turbidity	post turbidity	pre secchi depth	post secchi depth	pre TDS	post TDS	pre TSS	post TSS	pre Ca	post Ca	pre pH	post pH	pre tot. phos.	post tot. phos.	pre chlor a	post chlor a	fish community	pre Se	post Se	annual degree days growing season	year impounded	watershed size	runoff	sediment Hg
Smallwood	✓	✓	✓	✓	✓		✓		✓			✓	✓	✓			✓	✓	✓	✓				✓	✓	✓		✓			✓			✓	✓	✓	✓	✓
Ossokmanuan	✓	✓	✓	✓	✓		✓					✓	✓	✓				✓		✓				✓		✓		✓			✓			✓	✓	✓	✓	✓
Hinds Lake	✓	✓	✓	✓	✓		✓			✓	✓	✓	✓	✓	✓				✓		✓		✓		✓		✓				✓			✓	✓	✓	✓	✓
Cold Spring	✓	✓	✓	✓	✓	✓	✓	✓	✓	✓	✓	✓	✓	✓					✓				✓		✓		✓		✓		✓			✓	✓	✓	✓	
Great Burnt	✓	✓	✓	✓	✓	✓	✓	✓	✓	✓	✓	✓	✓	✓					✓				✓		✓		✓		✓		✓			✓	✓	✓	✓	
Cheticamp	✓	✓	✓	✓	✓	✓	✓		✓		✓	✓	✓		✓				✓	✓	✓	✓	✓			✓	✓	✓	✓	✓	✓		✓	✓	✓	✓	✓	✓
Gisborne	✓	✓	✓	✓	✓	✓	✓		✓		✓	✓	✓		✓				✓	✓	✓	✓	✓			✓	✓	✓	✓	✓	✓		✓	✓	✓	✓	✓	✓
McMillan	✓	✓	✓	✓	✓	✓	✓	✓	✓	✓	✓	✓	✓							✓		✓				✓		✓		✓	✓		✓	✓	✓	✓	✓	✓
Wreck Cove	✓	✓	✓	✓	✓	✓	✓	✓	✓	✓	✓	✓	✓		✓				✓	✓	✓	✓	✓			✓	✓	✓	✓	✓	✓		✓	✓	✓	✓	✓	✓
Long	✓	✓	✓	✓	✓		✓		✓		✓	✓	✓	✓						✓		✓		✓		✓		✓		✓	✓			✓	✓	✓	✓	✓
Outardes 2		✓		✓			✓		✓		✓	✓								✓						✓		✓		✓	✓			✓	✓	✓	✓	✓
Manic 1		✓	✓		✓	✓						✓	✓							✓						✓					✓			✓	✓	✓	✓	✓
Manic 3		✓	✓		✓	✓						✓								✓						✓					✓			✓	✓	✓	✓	✓
Manic 5		✓		✓	✓		✓				✓	✓					✓		✓				✓		✓		✓				✓			✓	✓		✓	✓
Opinaca	✓	✓		✓	✓		✓		✓		✓	✓				✓	✓	✓	✓		✓			✓	✓	✓	✓	✓		✓				✓	✓	✓	✓	✓
La Grande 2	✓	✓	✓	✓	✓		✓		✓		✓	✓				✓		✓	✓	✓			✓		✓	✓	✓	✓		✓				✓	✓	✓	✓	✓
La Grande 3	✓	✓		✓	✓		✓		✓			✓						✓					✓	✓		✓					✓			✓	✓	✓	✓	✓
Caniapiscau		✓		✓	✓		✓		✓			✓					✓		✓				✓	✓		✓			✓		✓			✓	✓	✓	✓	✓
Aubrey	✓		✓	✓	✓		✓		✓		✓	✓	✓				✓		✓							✓					✓			✓	✓	✓	✓	✓
Tunnel	✓		✓	✓	✓		✓		✓		✓	✓	✓				✓		✓							✓					✓			✓	✓	✓	✓	✓
Red Rock	✓		✓	✓	✓		✓		✓		✓	✓	✓				✓		✓							✓					✓			✓	✓	✓	✓	✓
Dog	✓		✓		✓				✓		✓	✓	✓				✓		✓							✓					✓			✓	✓	✓	✓	
SIL	✓	✓	✓	✓	✓	✓	✓	✓	✓			✓	✓	✓	✓	✓	✓	✓	✓	✓	✓	✓	✓	✓	✓	✓	✓	✓	✓	✓	✓			✓	✓	✓	✓	
Notigi	✓	✓	✓	✓	✓	✓	✓	✓	✓	✓	✓	✓	✓	✓			✓	✓	✓	✓	✓	✓	✓	✓	✓	✓	✓	✓	✓	✓	✓			✓	✓	✓		✓
Playgreen	✓	✓	✓	✓	✓			✓	✓	✓	✓	✓	✓	✓	✓				✓	✓	✓	✓	✓	✓	✓	✓					✓			✓	✓	✓		✓
Cedar	✓	✓	✓	✓	✓		✓			✓	✓	✓	✓	✓		✓				✓		✓		✓		✓					✓			✓	✓	✓		✓
Cookson	✓	✓	✓	✓	✓	✓	✓	✓	✓	✓	✓	✓	✓	✓					✓	✓	✓	✓	✓	✓	✓	✓	✓	✓	✓	✓	✓	✓	✓	✓	✓	✓		✓
Brazeau	✓	✓	✓	✓	✓		✓				✓	✓	✓						✓				✓		✓						✓			✓	✓	✓		✓
Williston	✓	✓	✓	✓	✓	✓	✓	✓	✓	✓	✓	✓	✓	✓	✓	✓		✓	✓	✓	✓		✓	✓	✓	✓	✓	✓		✓	✓			✓	✓	✓	✓	✓
Arrow	✓	✓	✓	✓	✓		✓		✓		✓	✓	✓	✓		✓	✓		✓		✓		✓		✓		✓		✓	✓	✓			✓	✓	✓	✓	✓
Libby	✓	✓	✓		✓		✓		✓	✓		✓					✓		✓							✓		✓		✓	✓			✓	✓	✓	✓	✓

* Sources of this data are given in Appendix C.

Table 23

TOTAL NUMBER OF FISH MERCURY SAMPLES AND SEDIMENT SAMPLES IN THE MASTER DATABASE,
PLUS THE NUMBER OF YEARS FOR WHICH DATA IS AVAILABLE FOR EACH OF THE MAJOR FISH SPECIES

Reservoir	Brook Trout	Lake Trout	Atlantic Salmon	Lake Whitefish	Mountain Whitefish	Northern Pike	Walleye	White Sucker	Longnose Sucker	Largescale Sucker	# Sediment Samples
Smallwood	89/1	353/1	5/1	678/1		175/1		61/1	162/1		19
Ossokmanuan		35/1	3/1	187/1		184/1		1/1	24/1		12
Hinds Lake	73/4		109/4								3
Cold Spring	43/2		78/2								
Great Burnt	29/2		72/2								
Cheticamp	10/1										3
Gisborne	10/1										3
McMillan	10/1										3
Wreck Cove	10/1										3
Long	4/1	10/1						10/1			3
Outardes 2				25/1		14/1		68/1			
Manic 1				2/1		15/1		30/1			
Manic 3				2/1		2/1		2/1		2/1	
Manic 5				7/1		6/1					
Opinaca				54/2		16/1	55/2		8/1		
La Grande 2				173/3		140/2	172/3		140/2		
La Grande 3				30/1		30/1	30/1		30/1		
Caniapiscau		106/1		137/1		57/1			114/1		
Aubrey						26/2	19/1	10/1			3
Tunnel				1/1		27/2	35/2	10/1			3
Red Rock						23/2	12/3	10/1			3
Dog						25/1					
SIL				175/7		175/7	158/8		100/4		12
Notigi				56/3		65/4	79/5				
Playgreen				25/1		30/2	35/2	25/1			
Cedar						165/10	116/9				
Cookson							99/3	50/2			
Brazeau						17/1		10/1			3
Williston				52/1				5/1			
Arrow			9/2						7/1	14/2	3
Libby			3/1							10/1	3

7 ANALYSIS OF EXISTING AND FIELD DATA

In the preceding section, we described the incorporation of data on mercury in fish collected from hydroelectric reservoirs, and on a variety of reservoir site parameters, into a single database for the purpose of statistical analysis. These data were derived from both existing datasets and from our own field data collections. Here we present the results of our analysis of these data.

This section is divided into three parts. First, we discuss our efforts to standardize the data on fish mercury levels so as to facilitate comparisons among reservoirs. Second, we consider the relationship between reservoir age and fish mercury levels in reservoirs for which more than one year's data is available on fish mercury. Third, we present the results of statistical analyses which evaluate the relative importance of the various factors hypothesized to influence mercury levels in fish (cf. Section 3.2 - Workshop Results).

7.1 STANDARDIZATION OF FISH MERCURY DATA

Before it is possible to compare the mercury levels in fish among several reservoirs, or through time in a single reservoir, it is necessary to account for differences in mercury levels which can be attributed to age or species differences among the samples used for comparison. The average mercury levels for all fish sampled from one reservoir may be higher than those for another simply because the fish sampled from the former are, on the average, either older, or from higher trophic levels. The effects of age and position in the food chain on body burdens are well documented (MacCrimmon et al., 1983; Wren et al., 1983).

To correct the data for fish age effects, one needs to answer the question, "What would the mercury level be in a fish of age 'i' if it was actually of some reference age 'j'?". By obtaining an answer to this question, one can convert the mercury data for all fish in all reservoirs into a corrected form that accounts for the effect, if any, of age. This

correction will remove any potential biases introduced by the differences in the age distribution, among reservoirs, of a particular species. The procedure we use to accomplish this correction is much the same as that used to compare regression lines in conventional analysis of covariance (Snedecor and Cochran, 1967).

The correction is based on defining a relationship between age and mercury levels in fish. Since we are only interested (at this point) in correcting for fish age effects, not for species or reservoir effects, this relationship must be defined independently for each reservoir, and for each species present in that reservoir. The effect of reservoir age is also of interest, so separate corrections are performed for each year of sampling. If these "strata" (i.e., species, reservoir) were not considered independently, their association with fish mercury levels would be obscured by the correction procedure.

Unfortunately, data on fish ages are only available for a relatively small number of the samples in our database. On the other hand, length data are available for every record in the database. Therefore, we have chosen to use length, rather than age, as our basis for correction. This does not pose a problem since by performing the correction in individual species and reservoirs, we are not required to assume that the relationship between age and length is consistent across these strata (which it probably is not).

We have chosen to use a linear model to carry out this correction. The choice of model (eg., linear, exponential, geometric) is more or less arbitrary, unless one has a sound theoretical or empirical basis for selecting a particular form. Other authors have used a similar procedure to that described here (eg., Ontario Ministry of the Environment, 1977), but chose a geometric (log-log) model. Their rationale for choosing this model was that it provided the best fit to their mercury-length data. In our case, a linear model performed well whenever a significant relationship was observed.

For each species, a reference length was chosen (the same for all years and reservoirs - see Table 24) to provide a common standard for comparisons among reservoirs. These reference lengths were chosen such that all reservoir samples contained fish of that length. This ensured that the regressions computed for each reservoir always contained observations of the reference length, thus avoiding the dangers of extrapolating from a regression. The only instance in which this criterion was violated was for brook trout in Smallwood Reservoir in Labrador. Populations of brook trout in this area are noted for their unusual size and age structure (D. deGraaf, personal communication), having much larger and longer lived fish than are typically found elsewhere in eastern Canada. This reservoir was therefore not included in any brook trout comparisons reported below.

For each species, year and reservoir, the predicted mercury level for a fish of reference length was computed from the regression:

$$[\hat{Hg}]_{REF} = a_{j,k,\ell} \cdot L_{REF,j} + b_{j,k,\ell} \qquad [7.1]$$

where:

$[\hat{Hg}]_{REF}$ ≡ the predicted mercury concentration (mg/kg) for a fish of the reference length (L_{REF});

$L_{REF,j}$ ≡ reference length (cm) for species j; and

a,b ≡ regression slope and intercept for species j, year k and reservoir ℓ.

Then, the individual observations of fish mercury were corrected by assuming that the corrected [Hg] would deviate from $[Hg]_{REF}$ by the same absolute amount that the observed [Hg] deviated from the concentration predicted from the regression:

$$[Hg]_i^{CORR} = [\hat{Hg}]_{REF} + ([Hg]_i - [\hat{Hg}]_i) \qquad [7.2]$$

where:

$[Hg]_i^{CORR}$ ≡ mercury concentration for fish i, corrected for length;

$[Hg]_i$ ≡ observed mercury concentration of fish i; and

$[\hat{Hg}]_i$ ≡ mercury concentration predicted for a fish of length L_i.

Table 24

REFERENCE LENGTHS USED FOR CORRECTING THE FISH MERCURY DATA FOR SIZE EFFECTS

Species	Reference Length (cm)
Brook Trout	24.0
Lake Trout	55.0
Atlantic Salmon	30.0
Whitefishes	40.0
Northern Pike	62.5
Walleye	45.0
Suckers	41.0

The correction procedure is depicted graphically in Figure 9. Note that if there is no significant regression (i.e., slope = 0), the correction procedure will have no effect on the variability in fish mercury levels (since none is explained by length).

To summarize, the correction procedure was used to estimate the expected concentration of mercury in each fish in the database, if it was of length L_{REF}, rather than its actual measured length. It is important to recognize that this method of correction preserves the variability among individual fish that is not explained by the regression of mercury on length, while accounting for differences among fish that <u>are</u> explained by this regression. By computing the regression separately for each species, year of sampling, and reservoir, differences in the mercury-length relationship associated with <u>these</u> factors are preserved. Corrected mercury concentrations (ranges, means, and standard deviations) and the regression results for selected species are listed in Appendix B, by reservoir and year of sampling.

Correction for species differences among reservoirs poses a more difficult problem. The simplest solution would be to only consider species that are common to all reservoirs in the database. Unfortunately, the database is not blessed with a ubiquitous species, so this course of action was not open to use. The most widespread species is northern pike, which has been sampled in at least one year in 19 of the 31 reservoirs. Pike were thus chosen as our standard species for intercomparisons among reservoirs.

For the 12 reservoirs without pike, an alternative means of standardizing interreservoir comparisons would be desirable. Ideally, one would like to be able to define a relationship between pike mercury levels and levels in some other species, present in these 12 reservoirs, such that the mercury levels in the latter species could be corrected to represent levels that might be expected in pike, if they were present in these reservoirs. This might be accomplished by comparing pike mercury levels with those of another species in reservoirs where the two species coexist. The ratio

$$\textcircled{A} = [\hat{Hg}]_i - [Hg]_i$$

$$\textcircled{B} = [\hat{Hg}]_{REF} - [Hg]_i^{CORR}$$

CORRECTION assumes $\textcircled{A} = \textcircled{B}$ $\therefore$ $[Hg]_i^{CORR} = [\hat{Hg}]_{REF} - ([\hat{Hg}]_i - [Hg]_i)$

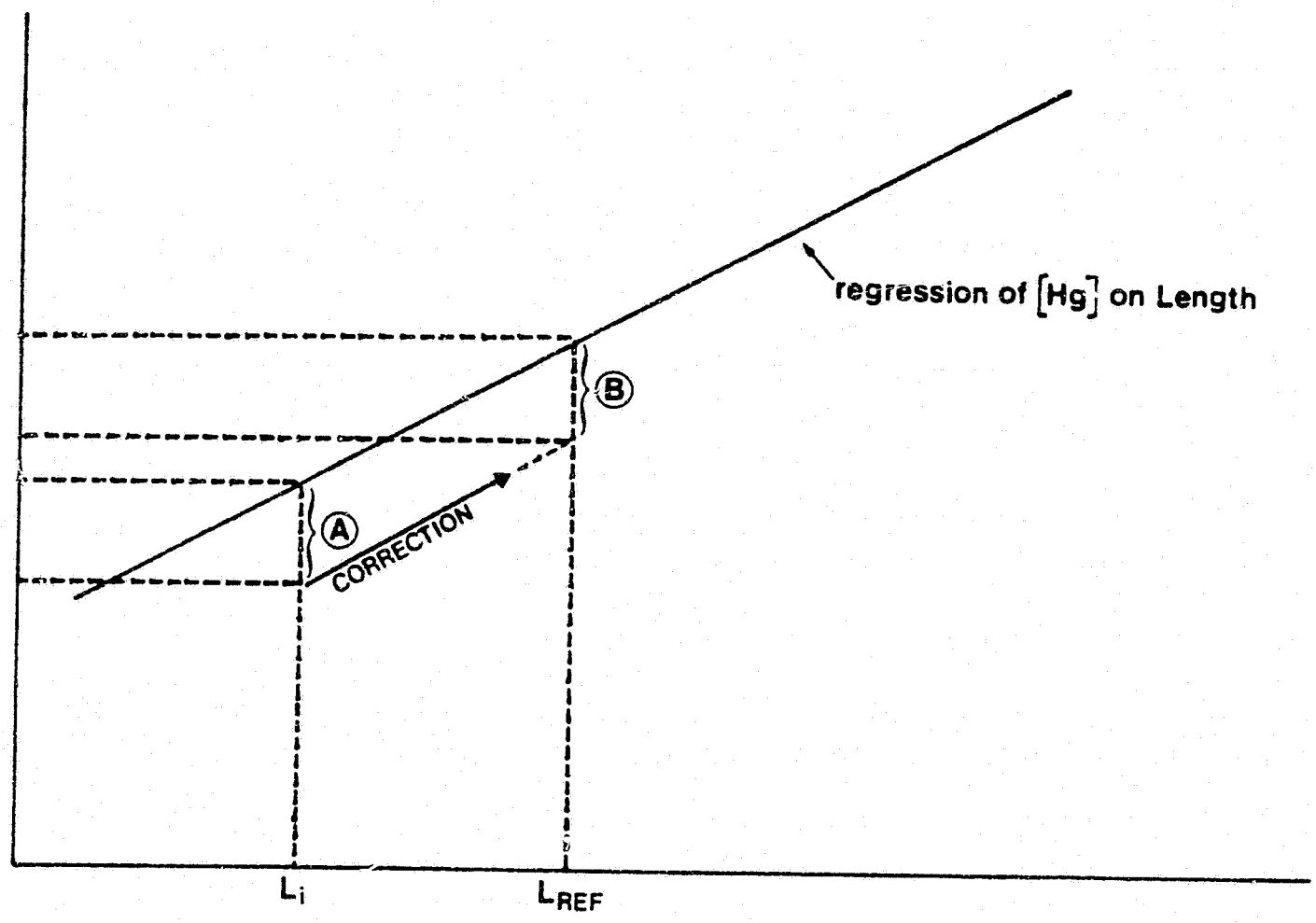


Figure 9 Graphical depiction of the correction of mercury data for fish size effects.

of the mercury levels for the two species (after accounting for size effects) could be used as a correction factor for fish of the other species in reservoirs where pike are not present:

$$[\hat{Hg}]^{A}_{i,PIKE} = \frac{[Hg]^{B}_{REF,PIKE}}{[Hg]^{B}_{REF,X}} \cdot [Hg]^{A}_{i,X} \qquad [7.3]$$

where:

$[\hat{Hg}]^{A}_{i}$ ≡ expected mercury concentration for fish i (as if it was a pike) in reservoir A (doesn't contain pike);

$[Hg]^{B}_{REF,PIKE}$ ≡ reference mercury concentration for pike in reservoir B (contains both pike and species X);

$[Hg]^{B}_{REF,X}$ ≡ reference mercury concentration for species X in reservoir B; and

$[Hg]^{A}_{i,X}$ ≡ observed mercury concentration for species X in reservoir A.

The critical assumption that underlies this correction procedure is that the relationship between mercury in pike and mercury in another species is the same for all reservoirs. To evaluate the validity of this assumption, we examined the variation in the ratio $[Hg]_{REF,PIKE}/[Hg]_{X}$, for several species which coexist with pike in some of the 19 reservoirs containing pike (Table 25).

It is readily apparent from Table 25 that while the expected trophic level differences are consistent across reservoirs (eg., pike levels are consistently above those of whitefishes and suckers), the variation in the ratio among reservoirs is considerable. For example, the ratio of pike mercury to whitefish mercury varies from 1.14 (Outardes 2) to 7.52 (Playgreen). This variability among reservoirs makes it necessary to reject the assumption that the ratio of mercury levels for two species is independent of the reservoir in which they are found, thus making it inappropriate to apply the species correction procedure described in Eq.

Table 25

COMPUTED VALUES OF $[Hg]^{B}_{REF,PIKE}/[Hg]^{B}_{REF,X}$
FOR EACH RESERVOIR CONTAINING BOTH PIKE AND SPECIES X

1. Whitefishes

Reservoir	Ratio	Reservoir	Ratio
Smallwood	3.13	La Grande 3	1.43
Ossokmanuan	3.00	Caniapiscau	1.50
Outardes 2	1.14	SIL	5.99
Manic Cinque	1.95	Notigi	6.21
La Grande 2	2.52	Playgreen	7.52

2. Suckers

Reservoir	Ratio	Reservoir	Ratio
Smallwood	3.61	Aubrey	2.91
Ossokmanuan	3.00	Tunnel	2.82
Outardes 2	1.12	Red Rock	4.24
Manic Un	4.48	SIL	9.35
Eastmain	2.09	Playgreen	3.75
La Grande 2	5.46	Brazeau	1.28
Caniapiscau	1.60		

3. Lake Trout

Reservoir	Ratio
Smallwood	1.122
Ossokmanuan	0.529
Caniapiscau	0.420

4. Walleye

Reservoir	Ratio	Reservoir	Ratio
La Grande 2	0.592	SIL	1.29
La Grande 3	0.543	Notigi	0.787
Aubrey	1.49	Playgreen	0.885
Tunnel	0.704	Cedar	1.33

7.3. Note, however, that the variation in the ratios within regions is generally much less than among regions. For example, if the 10 pike-whitefish ratios were ranked in ascending order, the five Quebec reservoirs would come first, followed by the two from Labrador, and finally the three from Manitoba.

To our knowledge, there is no other verifiable means of correcting for species differences among reservoirs. Thus, it is necessary to restrict the interreservoir comparisons to fewer than the entire set of 31 reservoirs. To allow incorporation into the analysis of reservoirs other than those containing pike, we have repeated the statistical comparisons described below for five other species or species groups: walleye, brook trout, Atlantic salmon, whitefishes, and suckers. While no single comparison includes all reservoirs, every reservoir is included in at least one comparison. Table 23 indicates which reservoirs contain each of the six species or species groups used in the analyses presented below.

7.2 RESERVOIR AGE

In our review of existing knowledge, it was noted that several authors have suggested that the phenomenon of elevated fish mercury levels in newly formed reservoirs is a transitory one (eg., Abernathy and Cumbie, 1977; Meister et al., 1979). This claim has been made despite the fact that there are no published time series of fish mercury levels in reservoirs which demonstrate both a rise in mercury immediately after impoundment and a subsequent decline as the reservoir ages. Our database does not contain such a time series, although the data from Manitoba (see Bodaly et al., 1984b for a more comprehensive discussion of these data) approaches this ideal. Nevertheless, we do have fish mercury levels measured in more than one year for 14 of the 31 reservoirs in the database, as well as data for the three reservoirs in Ontario (Aubrey Falls, Red Rock, Tunnel), which are in the same drainage and of differing age. In this section, we examine whether these data indicate significant changes in fish mercury levels with increasing reservoir age.

Of the 31 species-reservoir combinations we examined, 22 showed significant among-year differences (α = .05) when contrasted using non-parametric statistical tests (Table 26). Four of the nine non-significant contrasts were for the two insular Newfoundland reservoirs (Cold Spring, Great Burnt) for which there is only one year of post-impoundment data, the first year after impoundment. Of the remaining five, all but one (Red Rock, pike) are for fish exhibiting relatively low mercury levels (Figures 10 - 16). Thus, the data indicate a strong tendency for fish mercury levels to change as the reservoir ages.

In general, the younger reservoirs in the dataset, where significant changes were found, showed increases in fish mercury levels with age. Most striking are the results from Opinaca (Figure 11) and La Grande 2 (Figure 12), although the Hinds Lake Atlantic salmon (Figure 10) and the SIL lake whitefish (Figure 14) also show pronounced increases immediately after impoundment. Some of the older reservoirs show declines (Aubrey, Tunnel, Cedar), while for others, the pattern is less clear (Red Rock, Lower Arrow). For the reservoirs of intermediate age (7 - 10 years old), there is also some indication of decline (Figures 14 and 16), although the trend is quite variable for the two Manitoba reservoirs, especially Notigi. When all these reservoirs are considered together, the expected pattern of early increase followed by later declines is observed, although the timing of the decline may be highly variable (eg., 3 - 6 years for Cookson, 27 - 34 years for Tunnel). Nevertheless, Figures 10 - 16 demonstrate that for no reservoir in the database do we have a clear illustration of the complete sequence.

Comparisons of the mercury levels in pike in the three Ontario reservoirs sampled in 1984 also demonstrate significant differences among means (Table 27). Considered together, these three reservoirs do show the aforementioned pattern of increased concentrations followed by a decline, albeit on a much longer time scale than is apparent for other reservoirs in the dataset. The fact that significant declines through time were observed for both the youngest and oldest of these reservoirs (Figures 12 and 13), but not for the one of intermediate age, however, suggests that

Table 26

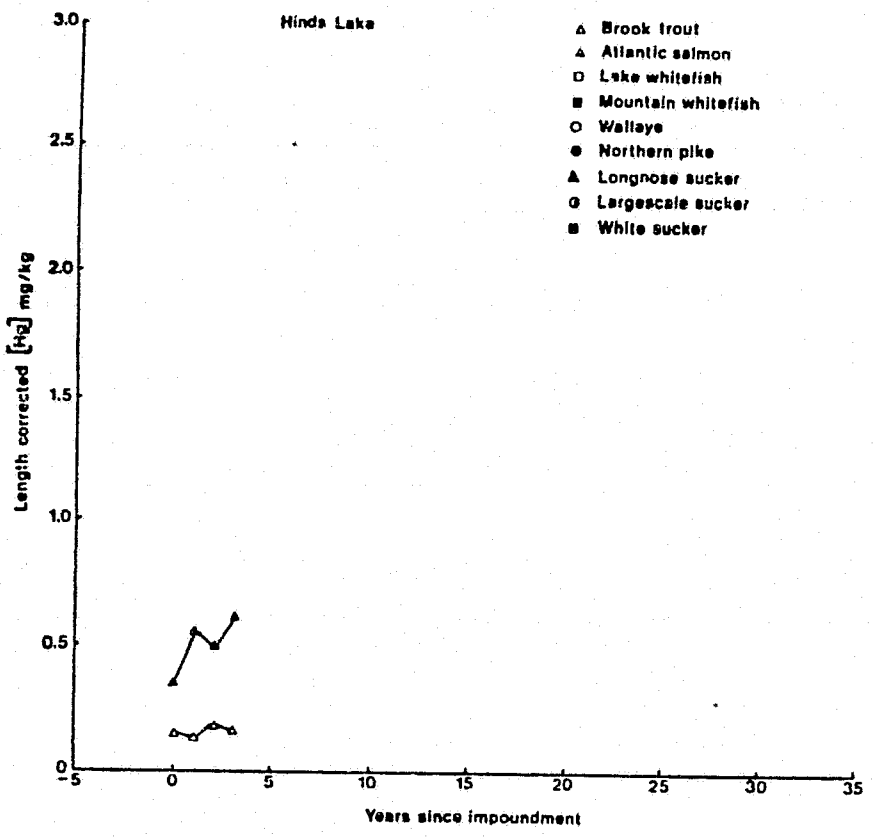
RESULTS OF STATISTICAL TESTS TO COMPARE FISH MERCURY LEVELS AMONG YEARS FOR FOURTEEN RESERVOIRS HAVING MORE THAN ONE YEAR'S DATA

Reservoir	Year of Impoundment	Species	No. of Years of Data	Range of Years	Probabilities of Among-Year Differences Arising by Chance
Hinds Lake	1980	Brook trout	4	80-83	.7065[1]
		Atlantic salmon	4	80-83	<.0001[1]**
Cold Spring	1982	Brook trout	2	82-83	.5179[2]
		Atlantic salmon	2	82-83	.5000[2]
Great Burnt	1982	Brook trout	2	82-83	.5268[2]
		Atlantic salmon	2	82-83	.4066[2]
Opinaca	1979	Lake whitefish	2	78-81	<.0001[2]**
		Walleye	2	78-81	<.0001[2]**
La Grande 2	1978	Lake whitefish	3	78-82	<.0001[1]**
		Walleye	3	78-82	<.0001[1]**
		Northern pike	2	78-82	<.0001[2]**
		Longnose sucker	2	78-82	<.0001[2]**
Aubrey	1969	Northern pike	2	77-84	.0005[2]**
Tunnel	1950	Northern pike	2	77-84	.0001[2]**
		Walleye	2	77-84	.0001[2]**
Red Rock	1960	Northern pike	2	77-84	.2779[2]
SIL	1976	Lake whitefish	7	75-83	<.0001[1]**
		Walleye	7	78-84	.0009[1]**
		Northern pike	7	78-84	.0082[1]**
		Longnose sucker	4	79-83	.1734[1]
Notigi	1976	Lake whitefish	3	80-83	<.0001[1]**
		Walleye	5	78-83	.0010[1]**
		Northern pike	4	80-83	.0001[1]**
Playgreen	1977	Walleye	2	78-81	.0228[2]**
		Northern pike	2	78-81	.1648[2]
Cedar	1965	Walleye	9	71-81	<.0001[1]**
		Northern pike	10	70-81	<.0001[1]**
Cookson	1977	Walleye	3	79-83	<.0001[1]**
		White sucker	2	80-83	.0210[2]**
Lower Arrow	1968	Mountain whitefish	2	80-84	.0079[2]**
		Largescale sucker	2	80-84	.7203[2]

1. Kruskal-Wallis statistic.

2. Median Test.

** Significant at the 5% level ($\alpha = .05$).



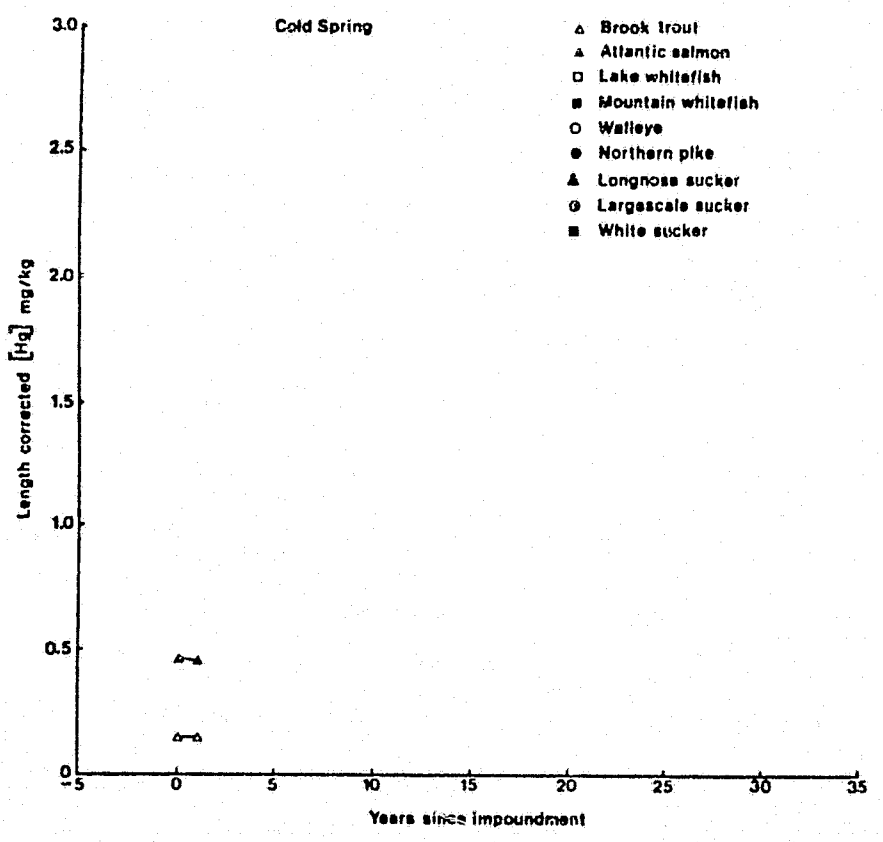


Figure 10 Changes in fish mercury concentrations with reservoir age: Hinds Lake and Cold Spring reservoirs.

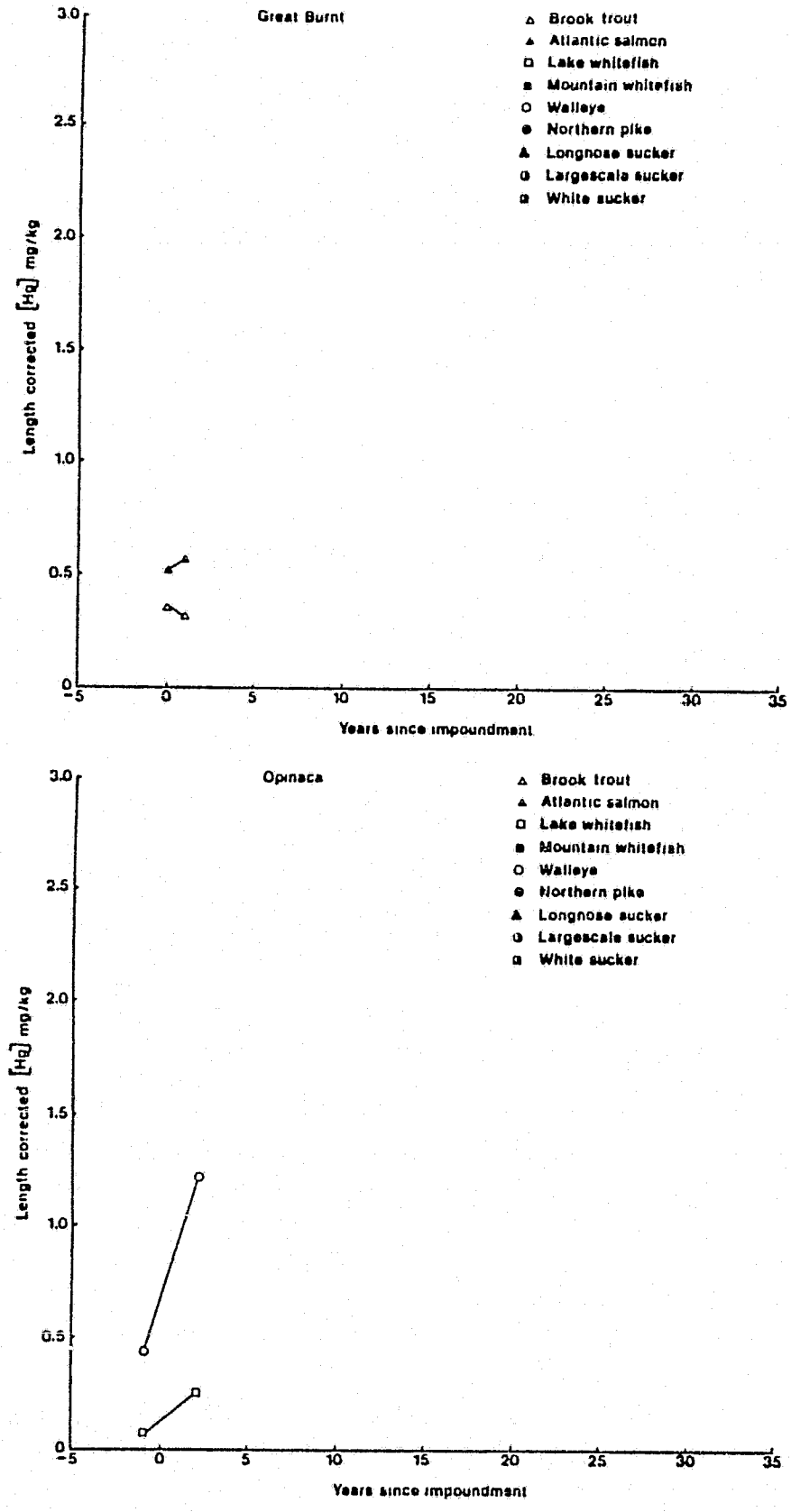
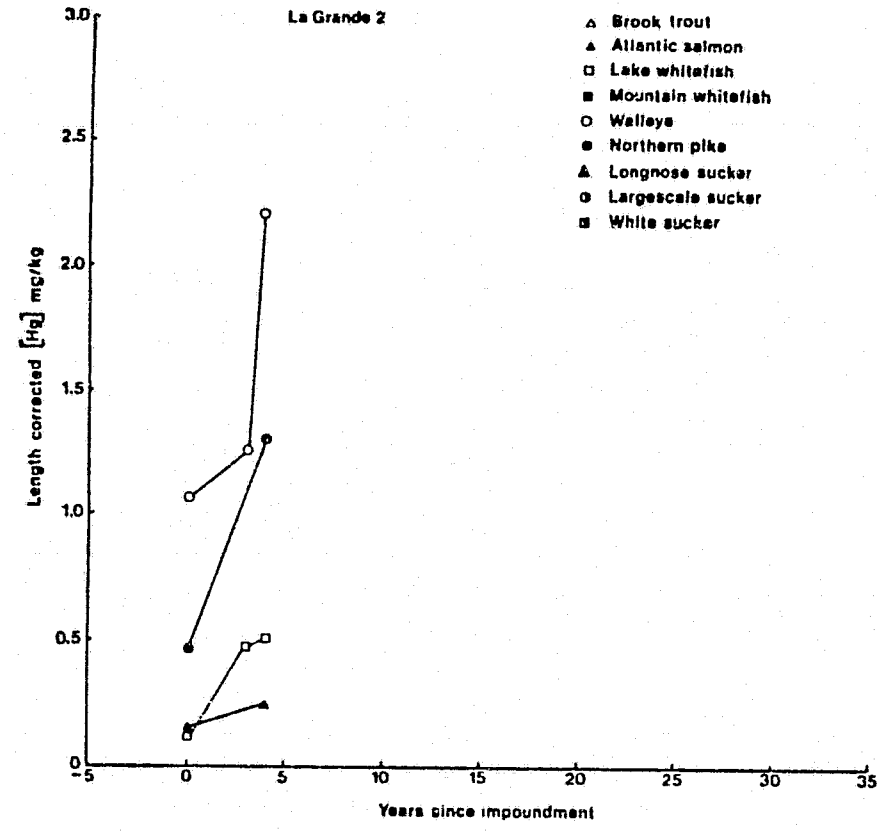


Figure 11 Changes in fish mercury concentrations with reservoir age: Great Burnt and Opinaca reservoirs.



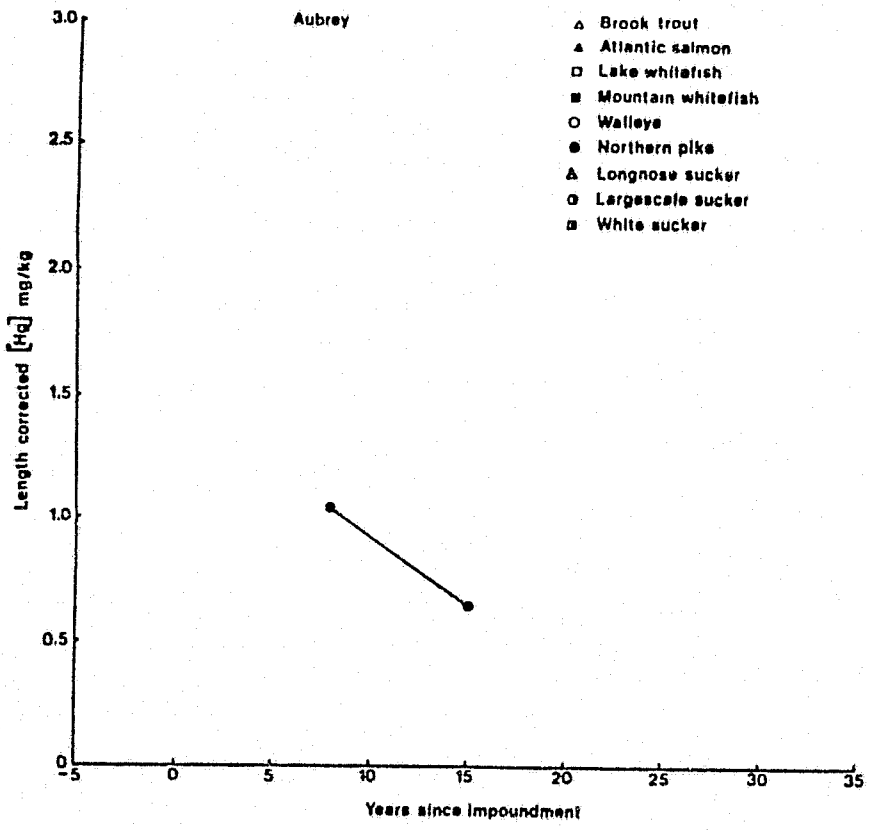
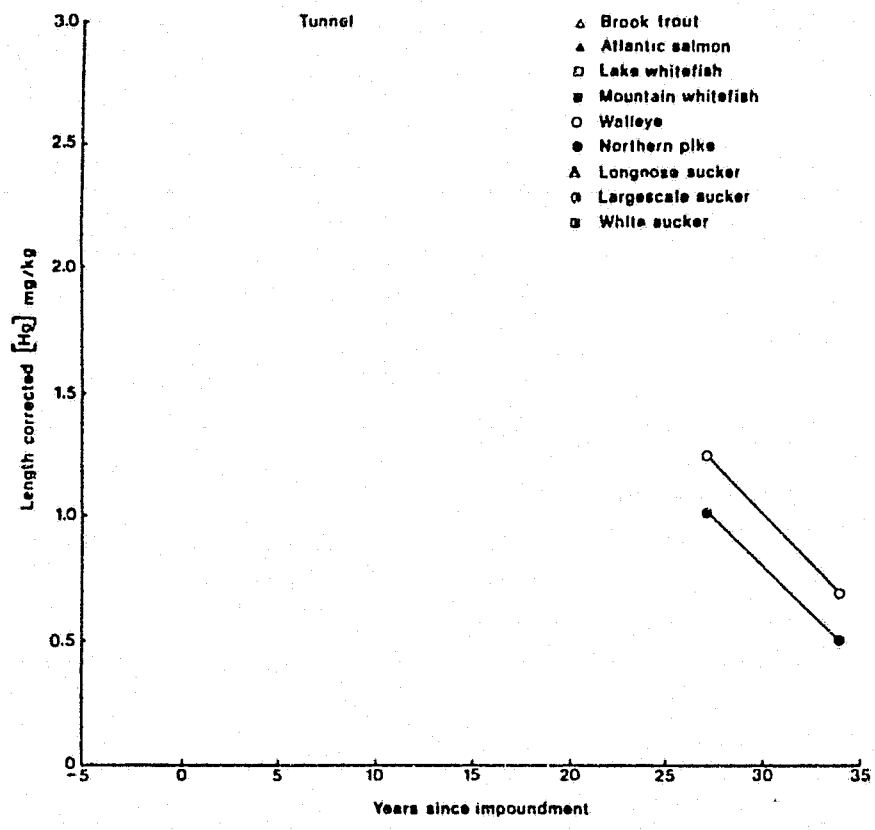


Figure 12 Changes in fish mercury concentrations with reservoir age: La Grande 2 and Aubrey reservoirs.



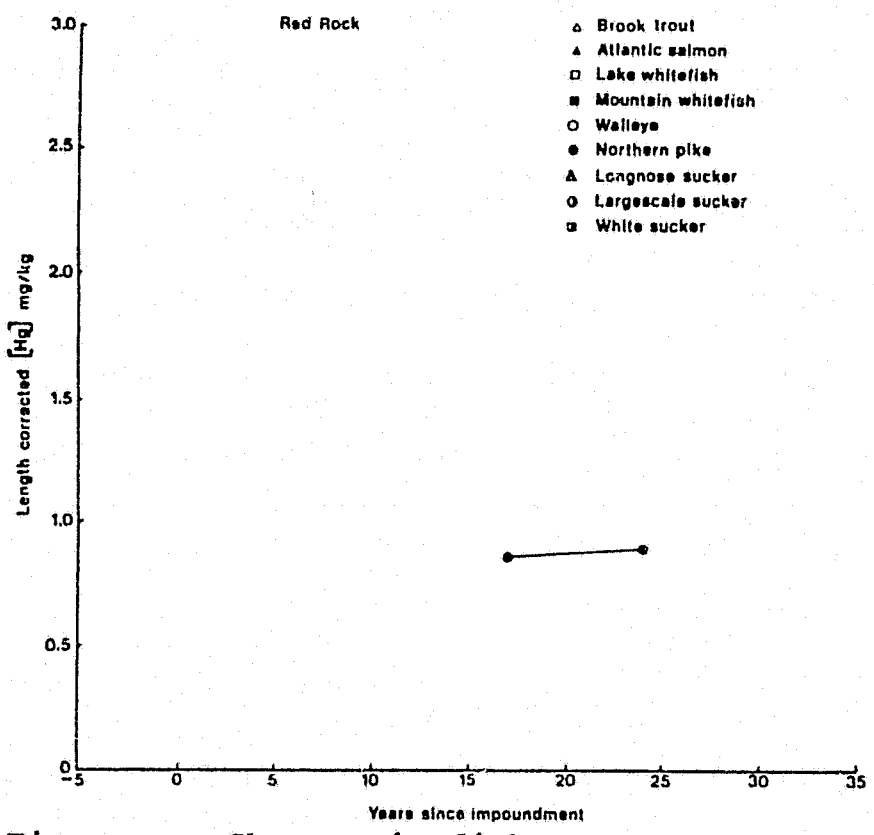
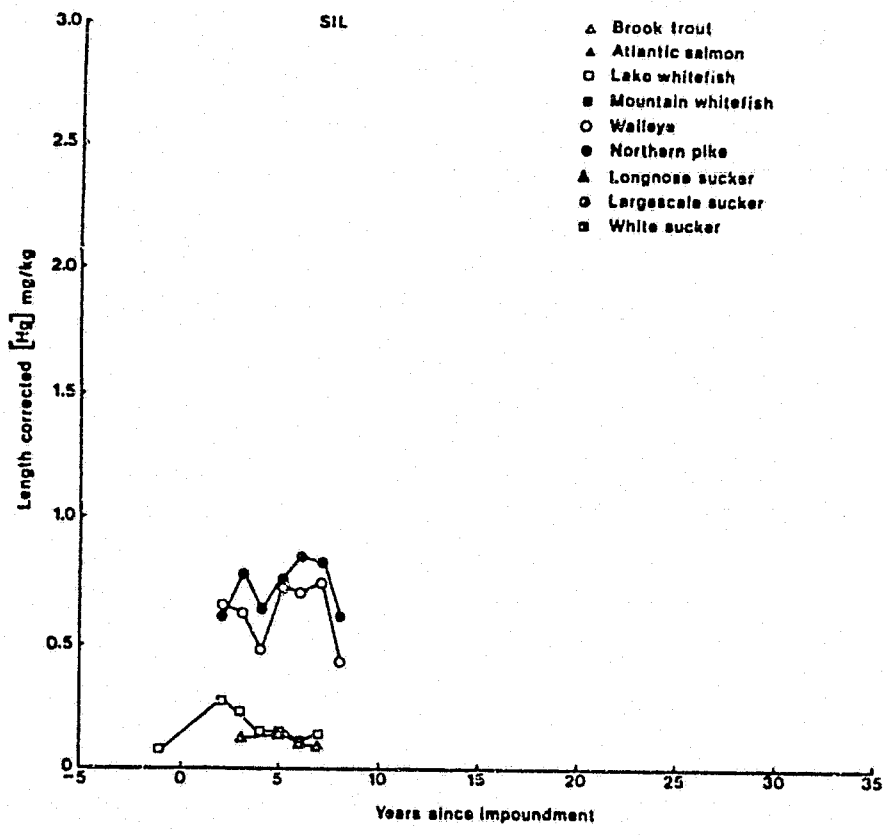


Figure 13 Changes in fish mercury concentrations with reservoir age: Tunnel and Red Rock reservoirs.



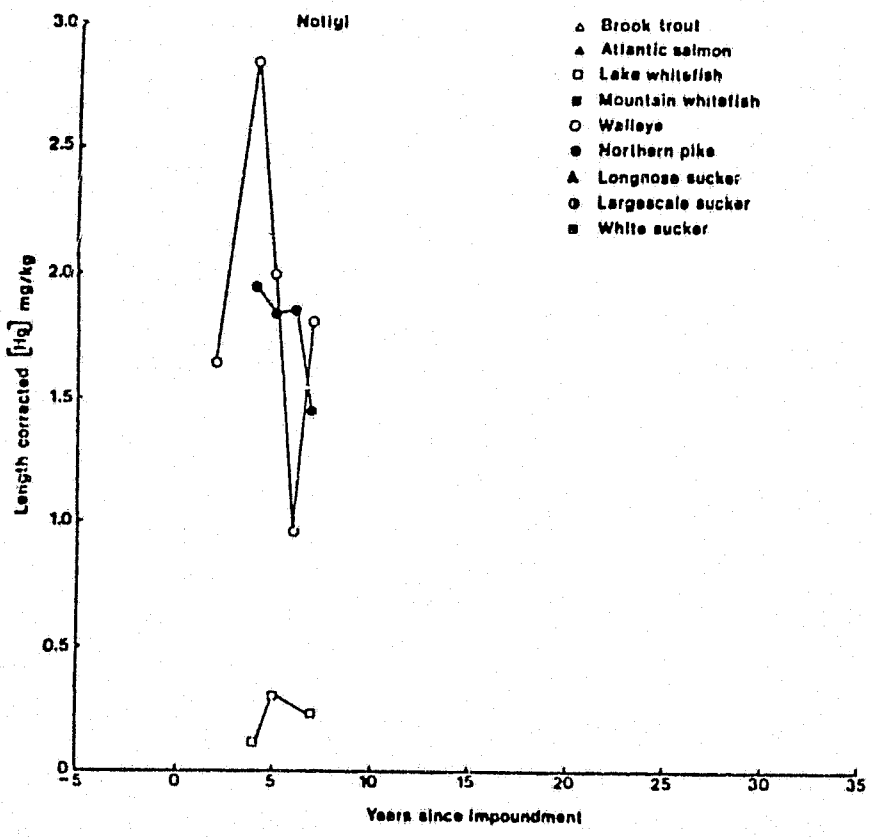
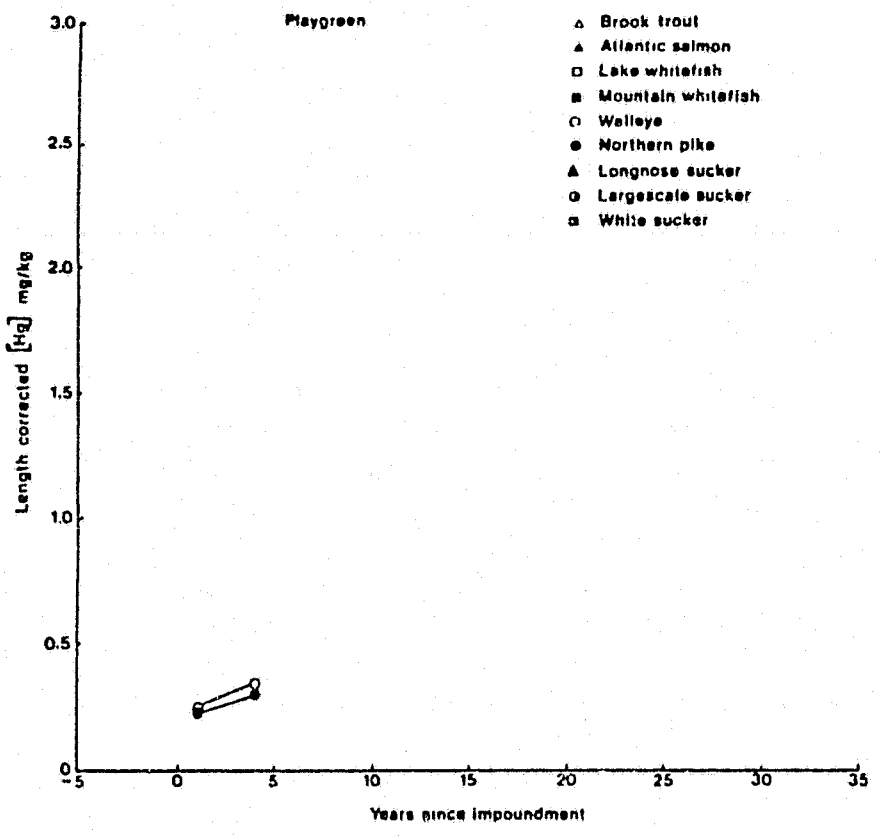


Figure 14 Changes in fish mercury concentrations with reservoir age: SIL and Notigi reservoirs.



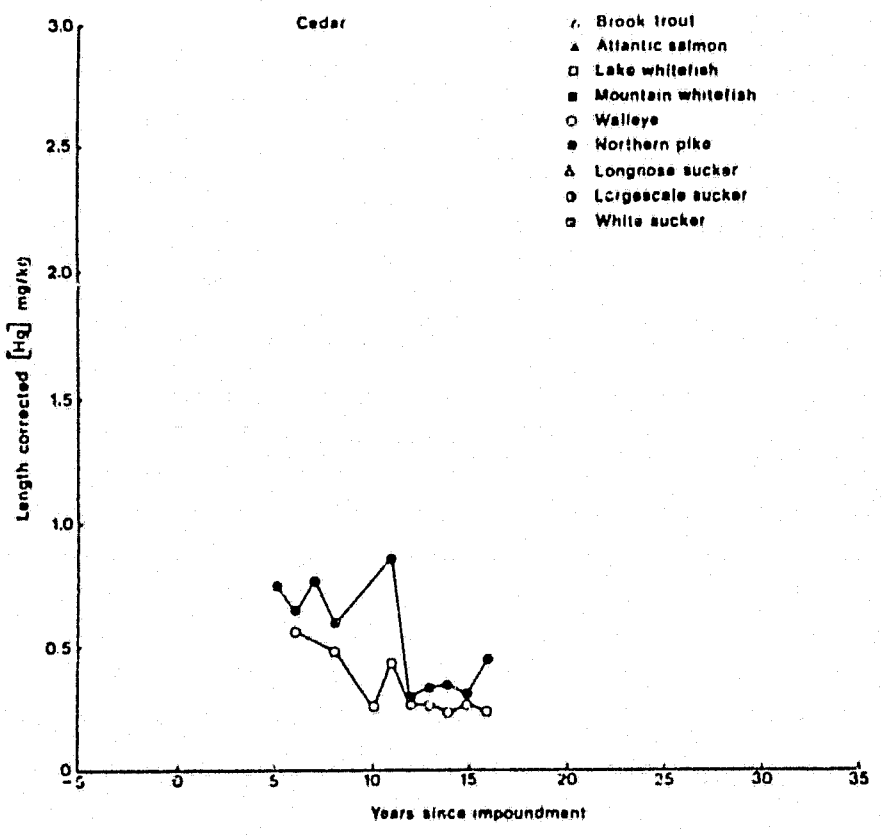


Figure 15 Changes in fish mercury concentrations with reservoir age: Playgreen and Cedar reservoirs.

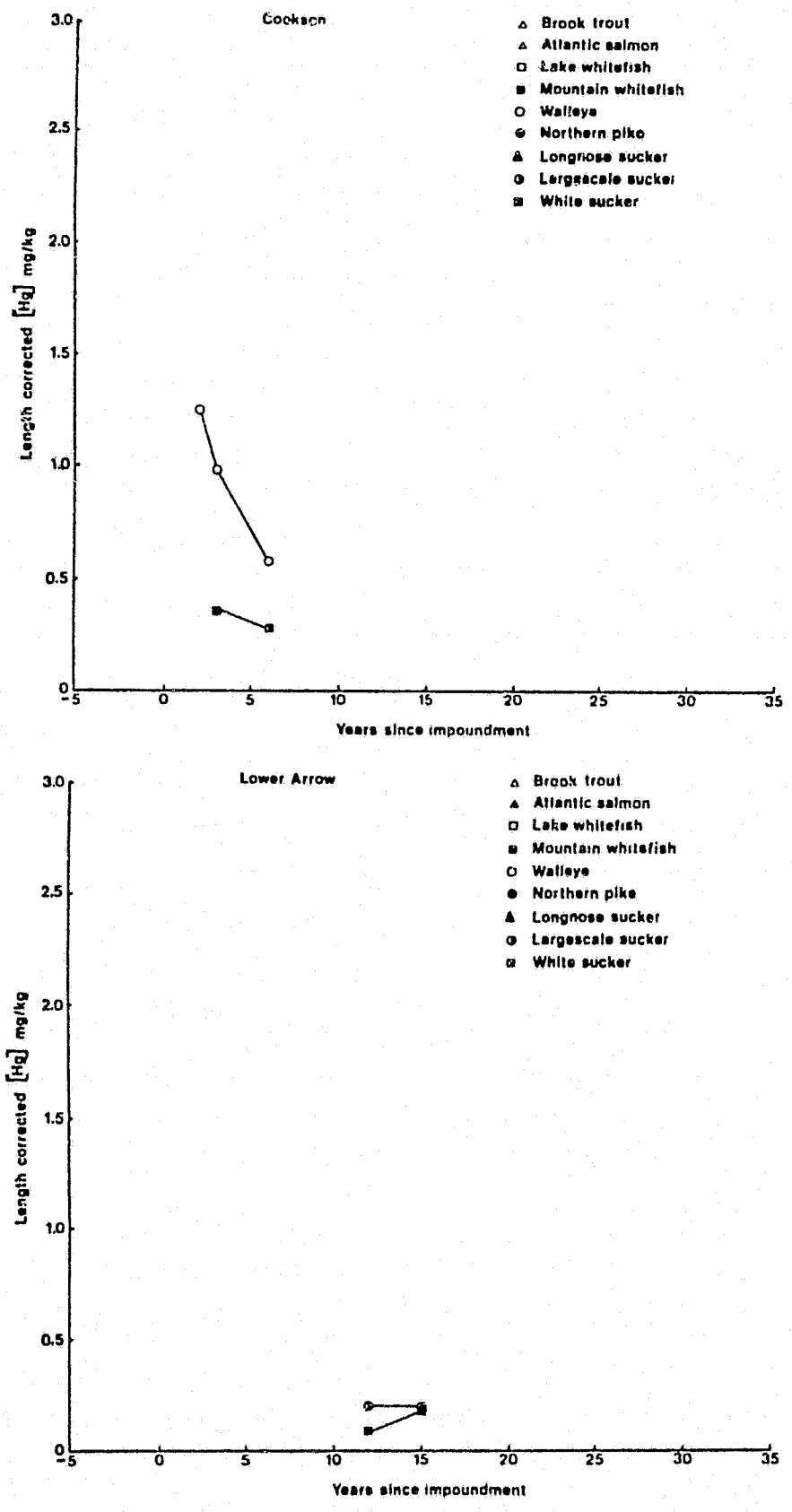


Figure 16 Changes in fish mercury concentrations with reservoir age: Cookson and Lower Arrow reservoirs.

Table 27

RESULTS OF STATISTICAL COMPARISONS OF MERCURY LEVELS
IN PIKE IN THE THREE ONTARIO RESERVOIRS SAMPLED IN 1984

Reservoir	Age at Time of Sampling	n	$\overline{[Hg]}$CORR (mg/kg)	Std. Error
Aubrey	15	11	0.64	.025
Red Rock	24	8	0.89	.046
Tunnel	35	12	0.48	.119

Kruskal-Wallis non-parameteric.

ANOVA on [Hg]CORR: K-W statistic = 15.01, $p < .001$, d.f. = 2

this result is perhaps spurious, rather than indicative of a pattern consistent across the three reservoirs. This result emphasizes the caution with which one must interpret interreservoir comparisons.

The demonstration of significant changes in fish mercury levels as reservoirs age presented a difficult problem for the statistical analyses intended to examine other factors which might affect mercury levels. Which year's data should be used to contrast the mercury levels between two reservoirs? If a consistent pattern with age was observed for all reservoirs having multi-year data, then a reservoir-age correction could be applied to all reservoirs in the same manner as the length correction applied earlier. The analysis presented above, however, suggests that such a consistent pattern does not exist among reservoirs. Rather than introducing a deliberate bias into the analysis by selecting data on the basis of an explicit criterion, such as choosing the year in which peak mercury levels were observed, we have opted to include all post-impoundment mercury data in the analysis, thereby ignoring the age effect. This action obviously raises the risk of results being confounded by reservoir age effects, but was considered the best of a number of options which would either deliberately bias the data or greatly reduce the opportunity for contrasts (by selecting only reservoirs with truly comparable data).

7.3 OTHER FACTORS

Before considering each of the individual factors which have been hypothesized to affect mercury levels, we conducted a more general statistical comparison of all reservoirs, which considered the effect of all factors simultaneously on fish mercury concentrations. To accomplish this we used a stepwise multiple regression technique, which assesses the relative contribution of each of the independent variables included in the analysis to explaining the observed variability in fish mercury concentrations. The analysis was conducted separately on each of the six species or species groups listed above, using length corrected mercury

concentrations for individual fish as the dependent variable.

Twenty-eight independent variables were used in this analysis (Table 28). All percentage data were transformed (arcsin square root) prior to analysis. The observed values for these variables, for each reservoir, are listed in Appendix C. Since values for most of these variables were not available for all reservoirs, it was necessary to use a dummy value wherever the actual value for a parameter was missing. The logical choice was to use the mean value observed for all reservoirs for which the parameter was available. This allows all reservoirs to be included in the analysis, without adding any information on the effect of a variable for reservoirs where none is available.

Tests of significance for this analysis rely on assumptions that all observations of the dependent variable are drawn from populations with normal distributions and similar variances. A preliminary examination of the data indicated that these assumptions are not valid, thereby casting doubt on such tests. Instead, it is more useful to simply examine which variables are included in the regression model and perhaps the order in which variables are added (Table 29), since this is based on the relative contribution they make to explaining the observed variability in fish mercury. Another indication of a variable's importance is given by the change in the r^2 value which occurs when the variable is added to the regression model. Note, however, that for independent variables which are correlated with each other, this change in r^2 can be affected by the order in which variables are added, since some of the variability in Y explained by variable X_1 may have been explained already by a correlated variable, X_2, added earlier in the regression procedure. This also implies that the order of addition of independent variables to the regression model should be interpreted with caution. Nevertheless, variables contributing the most to explaining the variance in the dependent variable, will generally appear first in the regression model.

The first result of note from these regressions is the value of the overall r^2 for each. For all species except Atlantic salmon (for which

Table 28

DESCRIPTIONS OF VARIABLES USED IN THE REGRESSION ANALYSES, ALONG WITH ACRONYMS USED FOR THESE VARIABLES IN SUBSEQUENT TABLES, AND ANY TRANSFORMATIONS COMPUTED PRIOR TO ANALYSIS

Variable	Acronym	Transformation	Reason
% rock	ROCK	arcsin $\sqrt{}$	percentage data
% sand	SAND	arcsin $\sqrt{}$	percentage data
% clay	CLAY	arcsin $\sqrt{}$	percentage data
% silt	SILT	arcsin $\sqrt{}$	percentage data
% other	OTHS	arcsin $\sqrt{}$	percentage data
% coniferous	CONI	arcsin $\sqrt{}$	percentage data
% deciduous	DECI	arcsin $\sqrt{}$	percentage data
% open vegetation	OPEN	arcsin $\sqrt{}$	percentage data
% cleared	CLEA	arcsin $\sqrt{}$	percentage data
fetch	FETCH		
relative area flooded[a]	RELA	arcsin $\sqrt{}$	percentage data
flooded area/ reservoir volume[b]	FLCONC		
post-impoundment volume	VOL.POST		
post-impoundment mean depth	MD.POST		
post-impoundment maximum depth	MXD.POST		
maximum drawdown	DRAWMX		
average drawdown	DRAWAV		
post-impoundment secchi depth	SEC.POST		
post-impoundment TDS	TDS.POST		
post-impoundment TSS	TSS.POST		
post-impoundment Ca	CA.POST		
post-impoundment pH	PH.POST		
post-impoundment phospherous	TP.POST		
post-impoundment chlor-a	CHLA.POST		
post-impoundment selenium	SE.POST		
total degree days	TEMP.TOT		
growing season d.d.	TEMP.GRO		
flushing time[c]	FLUSH		

a. Relative area flooded is calculated from (SA.POST - SA.PRE)/SA.POST, where SA.POST ≡ post-impoundment surface area; and SA.PRE ≡ pre-impoundment surface area.

b. Calculated from (SA.POST - SA.PRE)/VOL.POST.

c. Flushing time is calculated from Q/V, where Q ≡ runoff x drainage area; and V ≡ post-impoundment volume.

Table 29

RESULTS OF STEPWISE MULTIPLE REGRESSION ANALYSIS OF POST-IMPOUNDMENT MERCURY LEVELS IN 6 FISH SPECIES FOR 28 RESERVOIR SITE PARAMETERS

	Brook Trout	Atlantic Salmon	Northern Pike	Walleye	Whitefishes	Suckers
# Reservoirs	9	5	17	11	15	17
# Observations	253	181	1013	727	1343	618
Overall r^2	.507	.132	.557	.715	.464	.507
Variable included at: step 1 (r^2)	TDS.POST (.284)	PH.POST (.071)	CA.POST (.191)	MD.POST (.412)	CONI (.289)	DECI (.328)
2 (r^2)	SE.POST (.327)	FLUSH (.132)	RELA (.375)	FLUSH (.532)	RELA (.341)	RELA (.425)
3 (r^2)	MXD.POST (.378)		CLAY (.463)	CA.POST (.626)	DRAWAV (.411)	ROCK (.457)
4 (r^2)	CLAY (.484)		MD.POST (.531)	RELA (.692)	ROCK (.446)	SECCHI.P (.489)
5 (r^2)	DRAWAV (.507)		PH.POST (.540)	SAND (.707)	CLEA (.454)	TEMP.TOT (.492)
6 (r^2)			FETCH (.543)	TDS.POST (.713)	CHLA.POS (.459)	CONI (.499)
7 (r^2)			DECI (.547)	DECI (.715)	TSS.POST (.462)	FLCONC (.503)
8 (r^2)			DRAWMX (.550)		MD.POST (.464)	TDS.POST (.507)
9 (r^2)			DRAWAV (.554)			
10 (r^2)			TP.POST (.557)			

data are only available for 5 reservoirs), the final regression is able to account for more than 45% of the variability observed in the dependent variable. Given the expected natural variation in mercury concentrations in fish within a reservoir, these computed r^2 values are, in our view, remarkably high. The contrasts among reservoirs in both mercury levels and in at least some of the independent variables must be fairly great.

It is perhaps not surprising that no single independent variable consistently explains more variability in fish mercury levels than any other, for all species. This is almost certainly a reflection of the different set of reservoirs included in the analysis for each species. In other words, it is much more likely a statistical consequence of a limited (and varying) set of observations of the independent variables, than the result of real among-species differences in the factors that influence mercury accumulation. Nevertheless, this result is consistent with the view that mercury in fish in reservoirs is influenced by a number of factors, none of which dominate the process.

There are, however, two independent variables which do appear more prominent than the others. Relative area flooded, which was regarded as a key variable at the first workshop, appears to be important in four of the six regressions, although its incremental contribution to the overall r^2 is quite small. (It is noteworthy that the four Nova Scotia reservoirs, which only contain brook trout, did not show this effect; they were specifically chosen in this study to examine the importance of this factor.) Post-impoundment calcium, while only appearing in two regressions, is particularly important in both cases. A positive association was found in all instances for relative area flooded, while calcium was found to be negatively correlated with pike and walleye mercury. Both these results are consistent with expectation (see Section 2).

To further examine the contribution of individual factors to explaining among-reservoir variations in fish mercury concentrations we also carried out a series of regressions on single or small sets of variables (Table 30). Once again, the dependent variable used was length corrected fish

Table 30

RESULTS OF REGRESSIONS OF FISH MERCURY CONCENTRATIONS ON INDIVIDUAL VARIABLES OR SMALL SETS OF VARIABLES

Variables Included	Brook Trout			Atlantic Salmon			Northern Pike			Walleye			Whitefishes			Suckers			Factor being Examined
	# res	# obs	r^2	# res	# obs	r^2	# res	# obs	r^2	# res	# obs	r^2	# res	# obs	r^2	# res	# obs	r^2	
1. ROCK	9	253	.045	5	181	.048	14	1032	.002	11	727	.032	11	1339	.136	16	628	.117	1. soil type, erodability
2. CLAY & SILT	9	253	.043	5	-	-	14	1032	.000	11	727	.226	11	1339	.000	16	628	.086	1. soil type, bioturbation
3. CLEA	9	253	.018	5	181	.036	13	966	.031	8	664	.189	11	1371	.012	15	696	.004	2. vegetation type, clearing
4. OPEN	9	253	.009	5	181	.000	13	966	.012	8	664	.080	11	1371	.085	15	696	.002	2. vegetation type
5. DECI	9	253	.110	5	181	.023	13	966	.047	8	664	.121	11	1371	.216	15	696	.161	2. vegetation type
6. CONI	9	253	.149	5	181	.056	13	966	.015	8	664	.115	11	1371	.274	15	696	.071	2. vegetation type
7. FETCH, MD.POST, SAND, CLAY & SILT	8	-	-	3	-	-	9	636	.237	8	551	.585	8	1093	.413	12	564	.446	3. erosion rates
8. RELA	9	253	.041	5	181	.000	13	1013	.021	11	727	.106	11	1343	.135	15	618	.086	4. relative area flooded
9. FLCONC	9	253	.158	5	181	.004	12	983	.105	10	692	.176	10	1318	.052	15	593	.201	4. relative area flooded
10. SEC.POST	1	-	-	2	8	.408	11	796	.011	7	447	.012	10	1291	.020	11	503	.304	5. resuspension/sedimentation
11. TSS.POST	5	44	.074	0	-	-	4	435	.206	5	485	.048	4	240	.186	5	206	.083	5. resuspension/sedimentation
12. CA.POST	2	93	.052	2	8	.408	8	817	.254	5	485	.210	8	1164	.295	9	468	.108	7. Ca/pH
13. PH.POST	6	133	.006	2	8	.408	16	1037	.135	10	697	.327	13	1341	.242	17	696	.154	7. Ca/pH
14. TP.POST	6	133	.001	2	-	-	7	630	.083	5	404	.046	10	1235	.071	10	533	.000	8. trophic status
15. CHLA.POST	5	44	.015	0	-	-	4	330	.075	5	483	.091	7	419	.286	8	286	.108	8. trophic status
16. SE.POST	4	94	.012	0	-	-	0	-	-	1	-	-	0	-	-	1	-	-	10. Se
17. TEMP.TOT	9	253	.006	5	181	.039	17	1067	.132	11	727	.143	14	1371	.040	18	726	.006	11. temperature
18. TEMP.GRO	9	253	.002	5	181	.041	17	1067	.149	11	727	.190	14	1371	.074	18	726	.011	11. temperature
19. FLUSH	8	213	.142	4	138	.016	11	751	.003	7	398	.015	11	1265	.014	14	573	.004	13, 14. watershed size, runoff

res = # reservoirs

obs = # observations

\- = variance-covariance matrix singular

mercury concentrations for individual fish. For each of these regressions we only included those reservoirs for which all the variables in the regression were measured (i.e., no dummy values were used). Mean length corrected mercury concentrations for individual reservoirs are also plotted against individual factors for some of the more interesting results (Figures 17-20).

It must be emphasized that many of these variables can only be considered rather gross measures of the factors they are intended to represent. This is especially true for the soil and vegetation variables. Thus, it is probably wise only to attach significance to relatively strong associations (as measured by the r^2 parameter) between a variable or set of variables, and fish mercury concentrations. Such strong associations, while still not necessarily implying a cause-effect relationship, nevertheless point to empirical measures that might be used to anticipate mercury problems at proposed development sites. We have arbitrarily chosen a minimum r^2 value of 0.20 as our criterion for a "strong association".

Very few strong associations emerge from our analysis of the soil and vegetation variables. Mercury levels in walleye are positively correlated with % clay and silt, while whitefish mercury levels are higher in reservoirs for which relatively large areas of coniferous or deciduous vegetation were flooded. Since these associations are only found for a single species, however, it is probably best not to attach much significance to them.

The variables we assumed to be associated with erosion rates (fetch, mean depth, and % area in sand, clay, silt) collectively showed quite strong associations with mercury levels, for all species examined. The contribution of the individual variables to the regression r^2 varied considerably among species, however. Sand and clay were most important for the reservoirs containing pike, while fetch was important for whitefishes and suckers. For walleye, all five variables were of approximately equal importance. Worse, the sign of the association with a particular vari-

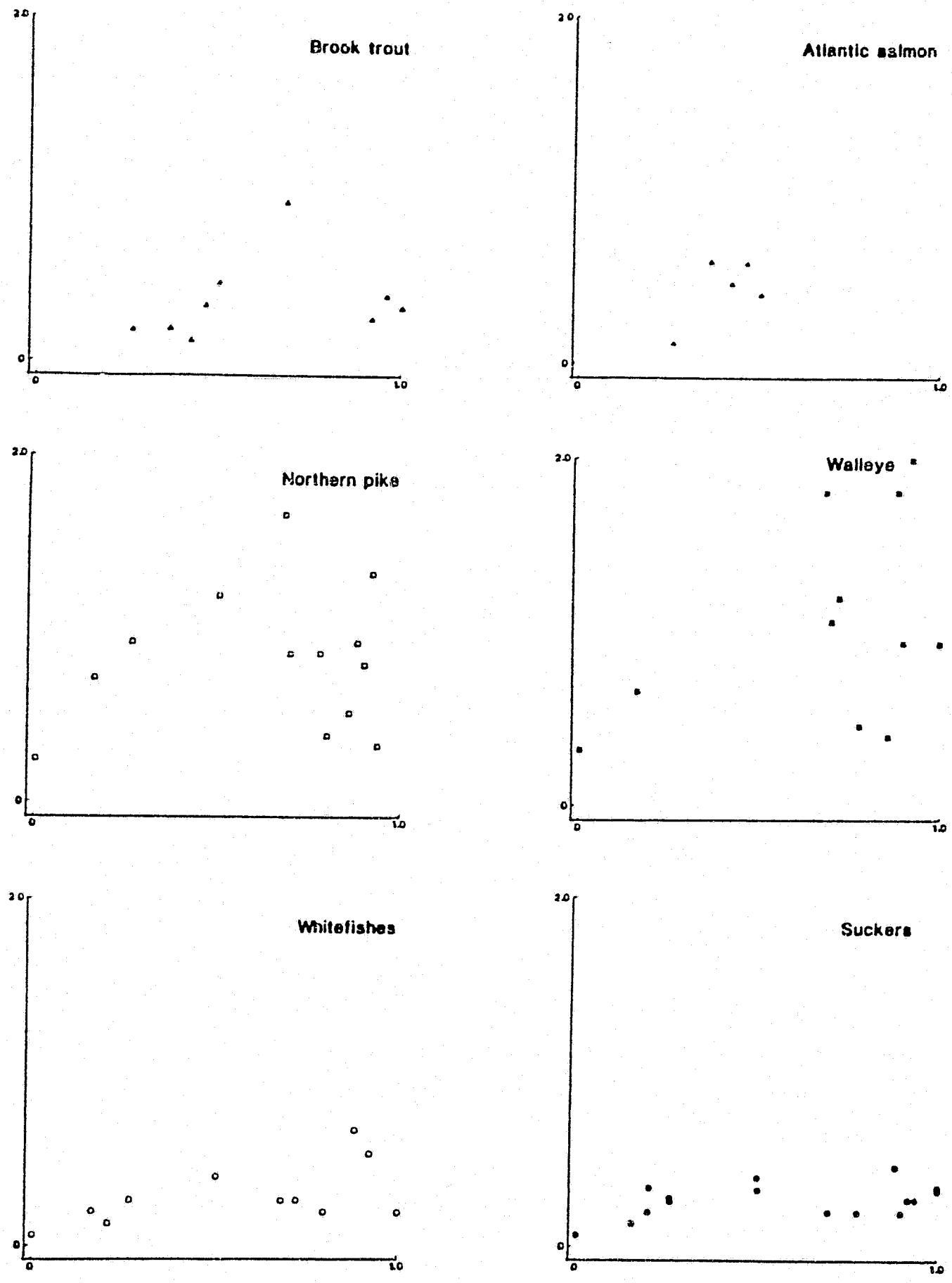


Figure 17 Relationship between relative area flooded and mean, length corrected fish mercury concentrations in reservoirs for six species or species groups.

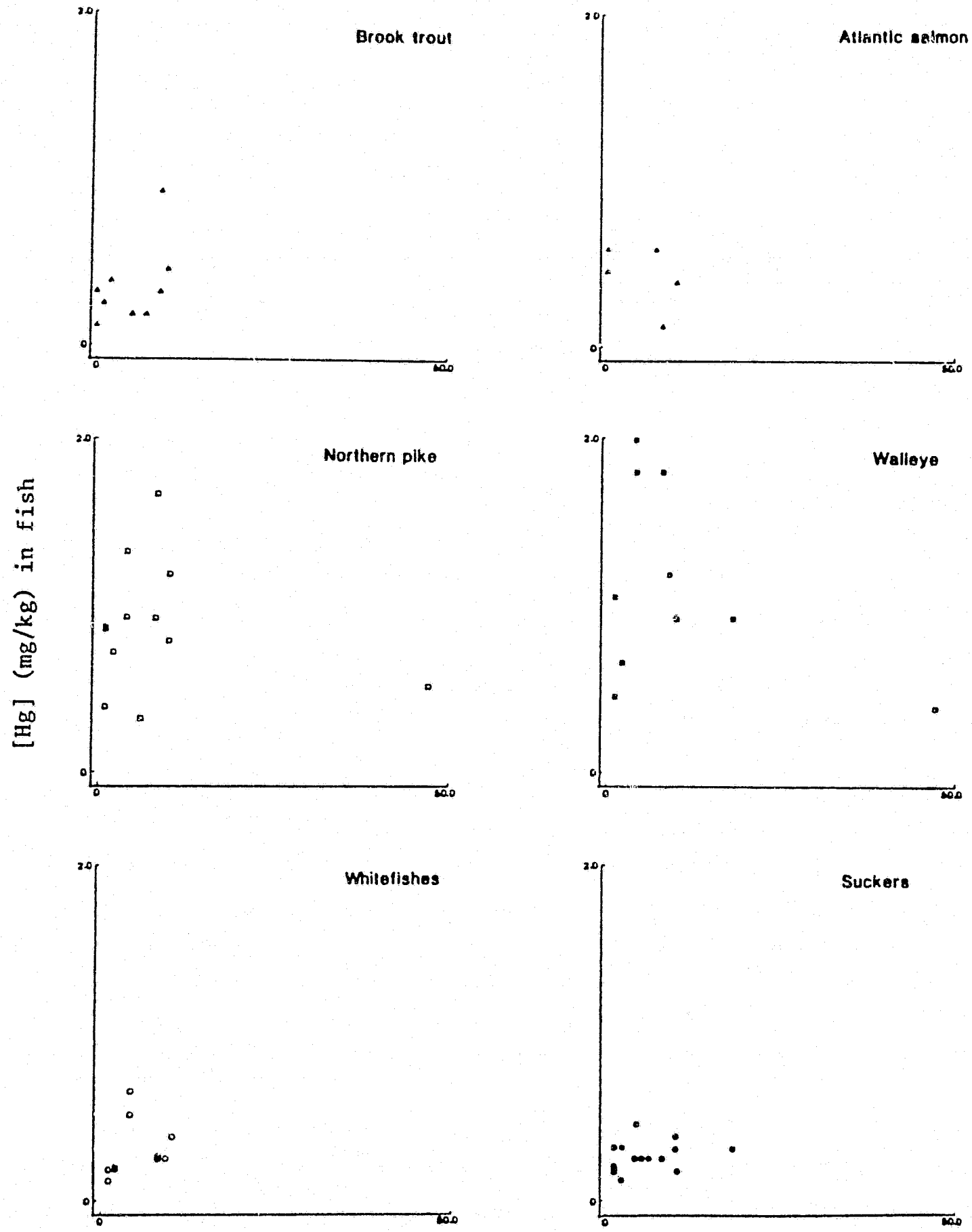


Figure 18 Relationship between area "concentration" (see Table 28) and mean, length corrected fish mercury concentrations in reservoirs for six species or species groups.

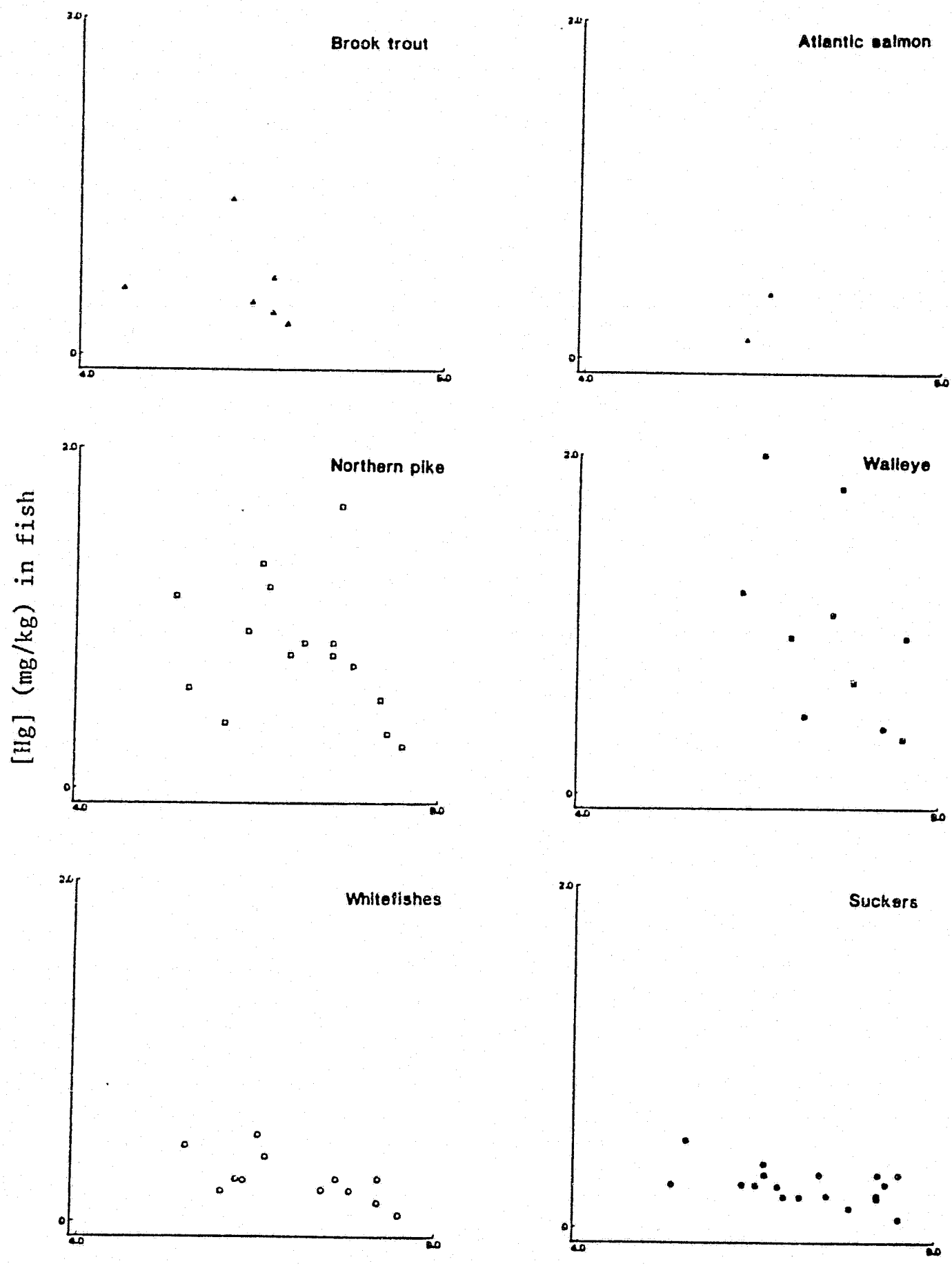


Figure 19 Relationship between reservoir pH and mean, length corrected fish mercury concentrations in reservoirs for six species or species groups.

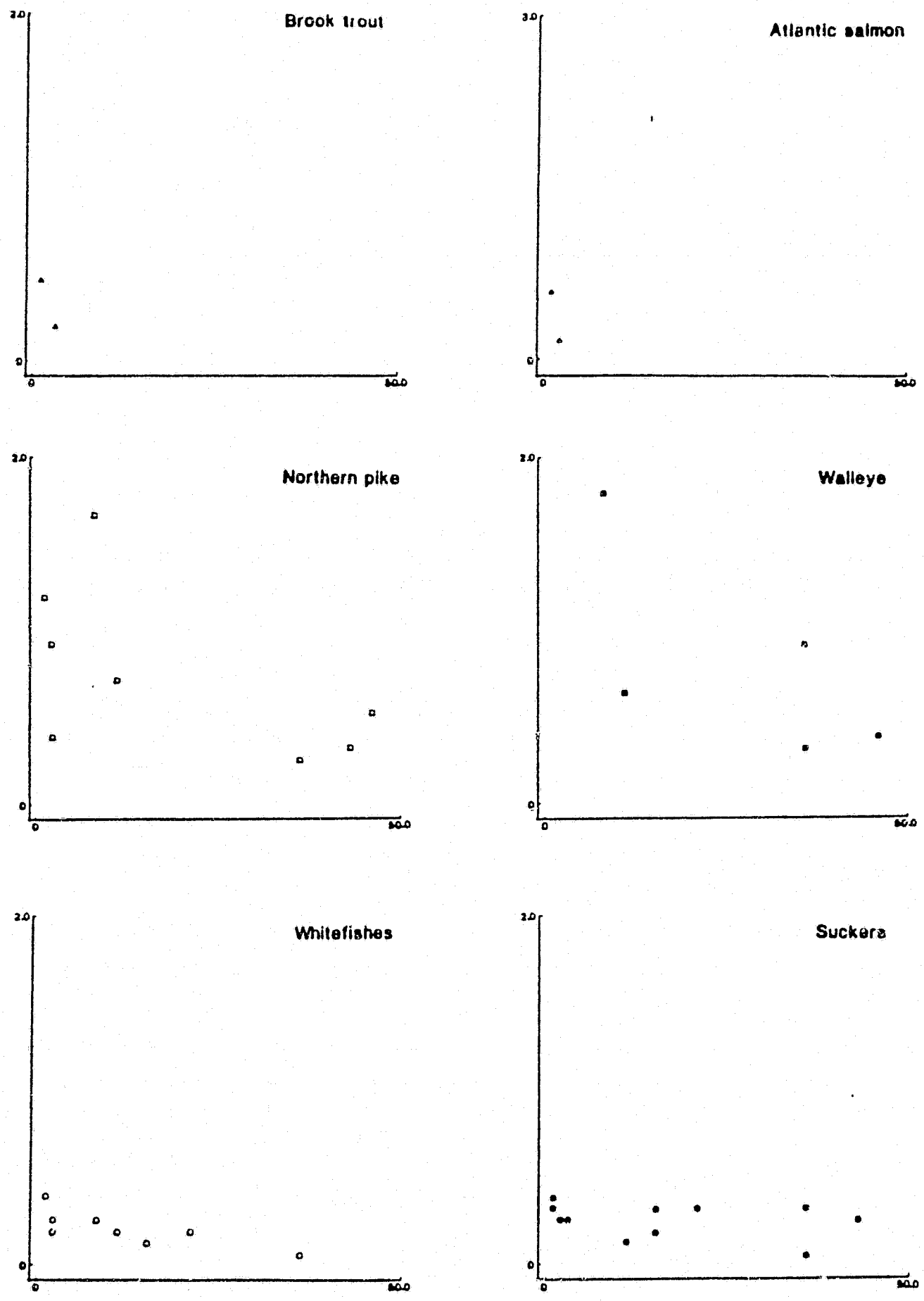


Figure 20 Relationship between reservoir calcium concentrations and mean, length corrected fish mercury concentrations in reservoirs for six species or species groups.

able differed among species (eg., fetch positive for whitefish, suckers; negative for pike, walleye), even when the variable was important in the regression. Since high erosion rates may actually stimulate mercury accumulation by supplying organic matter to the reservoir for methylation, or inhibit it by maintaining high levels of suspended inorganic matter, this result is not entirely surprising. The absence of a consistent pattern, however, makes it difficult to draw any meaningful conclusions about the actual importance of erosion from these data.

Relative area flooded was not strongly associated with mercury in this analysis. This is in contrast to the multiple regressions, although as mentioned earlier, the incremental contribution of this variable to the overall regression r^2 was quite small when other variables were also included. The weakness of this association is also evident from plots of mean mercury concentrations in fish versus relative area flooded (Figure 17). Inconsistent results were obtained for the second indicator of this factor (FLCONC), with a positive association emerging (as expected) for all species except pike and walleye. The results for the latter are dominated, however, by the extremely large value of FLCONC for Cedar reservoir relative to all the other reservoirs considered (Figure 18).

The variables associated with resuspension and sedimentation are also not strongly associated with mercury levels, except for isolated cases. For Atlantic salmon, the high r^2 for secchi depth includes data for only two reservoirs (8 observations), while the TSS regression for pike covers only four reservoirs, all in Manitoba. Nevertheless, the observed positive association with secchi depth (Atlantic salmon, suckers) and negative association with TSS (pike) are consistent with the hypothesis that suspended matter may act to reduce mercury uptake.

The most consistent pattern of strong associations occurs for the two variables representative of the cationic composition of the reservoir waters. Calcium in particular, but also pH, explain relatively high proportions (> 20%) of the total variability in fish mercury concentrations in 7 of 12 cases examined. In three other cases, the r^2 value exceeds

0.10. In addition, the sign of the association is consistent with expectation (negative) in all but one case. The single exception is pH for Atlantic salmon, for which only two reservoirs (8 observations) were included. Plots of calcium and pH versus fish mercury levels (Figures 19 and 20) illustrate this association, but also emphasize the high proportion of residual variability in mercury levels not explained by these variables.

None of the remaining variables show any consistent patterns of strong association with fish mercury concentrations. Chlorophyll-a was found to be negatively correlated to mercury in whitefishes, but this pattern was not repeated in any other species. The contrast in chlorophyll-a among reservoirs containing whitefish is very small (1 - 4 ug/L), and the r^2 is thus strongly influenced by the two Quebec reservoirs (Opinaca and La Grande 2) which have relatively high whitefish mercury levels, and low chlorophyll-a levels.

8 DISCUSSION AND CONCLUSIONS

In the introduction to this report we listed the six objectives identified by the CEA as the basis for this study, which were:

1) to identify sources and conditions most likely to result in mercury problems;

2) to summarize the state-of-the-art knowledge of mercury uptake processes;

3) to identify factors leading to greatest mercury bioaccumulation;

4) to develop methods for predicting possible mercury problems;

5) to provide guidelines for possible mitigative action; and

6) to recommend further research needs.

In this concluding section, we return specifically to these objectives (except #2, which was the explicit focus of Section 2 of this report). First, we consider objectives 1, 3 and 4, by examining how, on the basis of the results of this study and others reviewed earlier, one might assess the risk, or hazard of a mercury problem arising at a proposed reservoir site. Second, we consider the possible options for mitigation of mercury problems at sites where these problems have already been identified, or where they are expected to arise in the future (objective 5). Third, we identify those research needs which, in our view, have emerged from this study to be of greatest priority (objective 6). Finally we conclude our report with a few remarks on how this study has benefited from the use of the AEAM methodology.

8.1 ASSESSING THE RISK OF A RESERVOIR MERCURY PROBLEM

To develop a hazard rating scheme that is applicable in all regions of the country, one needs to consider two general classes of factors:

a) factors that change as a result of flooding, and consequently lead to elevated mercury levels relative to pre-impoundment conditions, or relative to nearby reference areas; and

b) factors that differ among reservoirs, and predispose certain areas or reservoirs to higher mercury levels in general or to a greater sensitivity to the factors in category (a).

Most other studies of the reservoir mercury problem, or of mercury bioaccumulation in general (i.e. those reported in our review of existing knowledge), have emphasized the first class, by examining individual, or small groups of systems, all in the same area. In contrast, the study reported here has concentrated on an empirical assessment of the second class, by drawing together information from throughout the country, and contrasting reservoirs from very different areas.

In our review of existing knowledge of the reservoir mercury problem, we emphasized the wide variety of factors that have been implicated as playing a role in determining post-impoundment increases in fish mercury levels. At the first workshop, this view was further stressed with the resulting recommendation that the focus of our research investigations should be on examining as many of these factors as possible, on as extensive a database as could be developed. While our analysis of individual factors did not identify a large number of parameters which explain significant amounts of the variation among reservoirs, the multiple regression analyses did demonstrate that a high proportion (> 45%) of the variation in mercury levels is explained by the complete set of variables. This result further supports the "multiple-factor" view of the mercury problem.

These observations indicate that while many factors appear to influence mercury levels, none of these factors consistently dominate, particularly when one compares reservoirs among broad geographical regions. Factors which are strongly associated with mercury levels in one region fail completely to explain variations in another area. A good example of this is provided by a comparison of the observations of Bodaly et al. (1984b)

with our results from the four Nova Scotia reservoirs. In flooded areas associated with the Churchill River diversion, Bodaly et al. (1984b) found a relationship between area flooded and mercury levels in predatory fish. In the Nova Scotia reservoirs, Wreck Cove Lakes had by far the highest mercury levels (.94 mg/kg in brook trout) and yet had the smallest area flooded relative to the original area of the lake.

Our analysis of the data from reservoirs across the country also provides some evidence for variations among species in the factors that influence mercury accumulation. This result must be interpreted with caution, however, since as pointed out earlier, differences in the range of physical and chemical characteristics across the set of reservoirs used for different species comparisons may be the primary reason for these apparent species differences. The small size of the existing database (in terms of the number of reservoirs included) is largely responsible for this result.

These considerations serve to emphasize what has to be the most important conclusion of this study. It is presently not possible to predict, with anything approaching a high degree of confidence, the likelihood of a mercury problem arising at a proposed site for hydroelectric development, especially when there is no previous experience in the same area. Extrapolations from one region to another are very likely to be misleading, and quite possibly grossly incorrect. It is very important to recognize this, and to accept that the mercury problem is typical of so many ecological problems, where much is often known about individual processes or specific sites, but attempts to generalize and predict have been, as often as not, unsuccessful. The complexity of this problem initially led us to abandon the idea of developing a simulation model. Now our uncertainty, especially on a regional basis, about the importance of factors which are presumed to play some role in the mercury accumulation process, forces us to conclude that even a more simple hazard rating scheme must only be applied with considerable caution. As discussed in Section 8.3, increasing the size and specificity of the database could increase the usefulness of such a hazard rating scheme.

While this must be the central conclusion of this study, it is equally important to stress that the knowledge that has been acquired through this study and others is nevertheless valuable. Bearing in mind the uncertainty which remains, this knowledge should be used to draw whatever general conclusions are possible about factors that should be assessed in evaluating the risk of a problem. The reservoir mercury problem is no less significant because we cannot predict its likelihood of occurrence accurately; the need to make the best use of the information that is available remains acute. On the basis of the analyses conducted in this study, and from other studies, a number of factors have emerged as being worthy of consideration in a risk assessment. These factors were discussed at length by participants at the second workshop (Table 31), and the results of these discussions are summarized below.

8.1.1 Reservoir Location

It is apparent from a comparison of the geographical distribution of reservoirs exhibiting high mercury levels to those with lower levels that certain areas appear to be predisposed to mercury problems. The reservoirs in northern Manitoba, northern Ontario, the James Bay area of Quebec, and Labrador generally show higher mercury levels than are found in reservoirs from Nova Scotia, New Brunswick, Southern Manitoba, Alberta and British Columbia. The pattern is less clear in insular Newfoundland, the Quebec north shore and southern Saskatchewan. These differences may in part be a reflection of species differences among reservoirs, and there are clear exceptions to the rule (e.g., Wreck Cove Lakes, Nova Scotia), but a general pattern of regional differences nevertheless does exist. This pattern may or may not be explained by other measurable factors such as geology or water quality, but even without such knowledge one can reasonably assign higher risk to reservoirs in areas noted for the problem.

As well as examining the observed mercury levels in other reservoirs near a proposed site, useful information can also be obtained from natural

Table 31

PARTICIPANTS AT THE SECOND MERCURY WORKSHOP

Name	Affiliation
Hugh Bain	Department of Fisheries and Oceans
Wayne Carey	Ontario Hydro
Gina Cunningham	ESSA Ltd.
Dirk deGraaf	LGL Ltd.
Gaetan Guertin	Hydro Quebec
Bob Hecky	Freshwater Institute
Mike Jones	ESSA Ltd.
David Marmorek	ESSA Ltd.
Bill Peel	Alberta Power Ltd.
Bob Stedwill	Saskatchewan Power Corp.
Pam Stokes	Institute for Environmental Studies
Terry Toner	Nova Scotia Power Corp.
Tom Watson	B.C. Hydro
Chris Wren	Institute for Environmental Studies

lakes in the area. Factors which predispose reservoirs to elevated mercury levels may well also give rise to high levels in natural water bodies. If lakes in the area are naturally high in mercury, it may be that high reservoir levels have nothing to do with the actual impoundment. Such information is obviously of considerable use to utilities proposing a new development, or facing claims for compensation at an existing site. On the other hand, it is also possible that in an area where natural mercury levels are high, even a small increase due to impoundment may lead to a much higher incidence of fish body burdens exceeding health standards.

Clearly, information from natural lakes would be of considerable value in evaluating either the risk of a problem, or the evidence that a problem has arisen due to impoundment (by comparing existing reservoirs to natural lakes). Some databases already exist which report fish mercury levels in natural lakes. One example of this is the National Inventory Survey of lakes and rivers conducted by the federal Department of Fisheries and Oceans to assess the current status of lakes at risk of damage due to acidic precipitation. The survey covered 814 lakes throughout eastern Canada, and one of the measured parameters in some of these lakes was fish mercury levels. In Ontario, 39 northern pike were sampled from eight different lakes, mostly in the north-central region (Sault Ste. Marie - Sudbury).[1] The mean length-corrected mercury concentration for these fish was 0.584 mg/kg (standard deviation = 0.34). This compares to the maximum values for each of the four Ontario reservoirs of 1.03 (Aubrey), 1.01 (Tunnel), 0.89 (Red Rock) and 0.77 (Dog). Further comparisons of this type, ideally on more restricted geographical scales, would obviously be useful as ingredients of either a risk assessment or a damage evaluation.

[1]These data were kindly made available to us by J.R.M. Kelso, Department of Fisheries and Oceans, Sault Ste. Marie, Ontario.

8.1.2 Water Hardness

Our inter-reservoir comparisons indicated that pH and calcium showed the strongest association of all variables considered with mercury levels in fish. Since only one reservoir in the database has a pH less than 5.0, this effect is more likely a result of differences in the overall ionic strength of the waters than a consequence of important variations in reservoir acidity. Several studies were discussed in our review of existing knowledge which indicated that uptake of metals by fish is higher in soft water than in hard water (e.g., Rodgers and Beamish, 1981; Spry et al., 1981; Zitko and Carson, 1976). As well, other studies have documented the tendency for mercury levels to be higher in lakes of lower pH (e.g., Suns et al., 1980; Wren and MacCrimmon, 1983). Finally, declines in surface water pH of 0.5 units (6.5 to 6.0) have been cited as contributing to post-impoundment increases in fish mercury in reservoirs in the James Bay development (Boucher and Schetagne, 1983).

These data all suggest that water hardness is an important factor in controlling mercury levels in reservoir fish. On the other hand, low pH is clearly not a necessary condition for elevated mercury levels, as evidenced by the high levels in SIL and Notigi reservoirs, both of which have circumneutral or slightly alkaline pH values. As well, the recent studies of Ramlal et al. (1985a) and Xun et al. (1985) on the effect of pH on methylation and demethylation in sediments and water illustrate that this association is by no means straightforward, since reduced pH appeared to stimulate methylation in the water column but had the opposite effect in sediments. Nevertheless, the widespread empirical support for the importance of this factor suggests that it deserves a prominent role in a risk assessment.

8.1.3 Extent of Flooding

In our analysis we considered two measures of this factor: 1) changes in reservoir surface area after impoundment relative to the system's original surface area (RELAFL); and 2) changes in surface area relative to the

reservoir's post-impoundment volume (FLCONC). Neither of these variables showed a particularly strong association with fish mercury levels in our analysis of individual factors, but the first did emerge as being relatively important in the multiple regression analyses. Other evidence is also conflicting. Bodaly et al. (1984b) cite this factor as being of central importance in the Churchill River diversion area in northern Manitoba, while our comparison of four Nova Scotia reservoirs and a similar evaluation of reservoirs in insular Newfoundland (H. Bain, personal communication) did not find such a pattern.

If the major mechanism by which impoundment leads to elevated mercury levels in fish is via the inundation of organic matter due to flooding, and consequent increases in bacterial methylation rates, then it stands to reason that this factor should be important. There is good evidence from mesocosm studies (Furutani and Rudd, 1980; R.E.Hecky, personal communication) that the supply of organic matter as a substrate for microbial methylation plays a major role in influencing mercury uptake rates by fish. That this factor is not prominent at all locations where it has been considered, however, suggests that other factors may often overwhelm its effect.

One factor which was suggested at the second workshop as possibly confounding the effect of the extent of flooding is the rate of flooding. If flooding occurs very rapidly, the opportunity for bacterial methylation to occur before the flooded area becomes inundated by cold, hypolimnetic waters (presumably less conducive to high rates of microbial activity) will be considerably reduced. Some reservoirs are flooded very rapidly (e.g., Revelstoke reservoir in B.C. was flooded in approximately 4 weeks; T.A. Watson, personal communication) while for others the flooding may take several years to complete. To our knowledge, this factor has not been explicitly considered in any study of reservoir mercury problems.

If the littoral zone is the primary area of methylation and mercury uptake, then another factor which may be very important is the morphometry of the reservoir after impoundment. Reservoirs with steep banks

or with relatively little shoreline development (defined as $L/(2*(\pi* A)^{\frac{1}{2}})$, where L is the shoreline length and A is the reservoir surface area) might be expected to show less of an effect of the extent of flooding than would other systems.

The most important factor which must be considered together with the extent of flooding, however, is the nature of the flooded areas. If these areas have little or no vegetation or organic soils, then the extent of flooding is unlikely to be important. In our analysis, we did not find that the percent area cleared was associated with mercury levels, but this is probably because our measure is not a very good one. For example, many of the reservoirs in our database whose watersheds were reported to have been extensively cleared had actually only had merchantable timber removed. The remaining vegetation in these areas may well have been more important than that which was removed.

In summary, the extent of flooding is clearly not a consistent, unequivocal indicator of the expected magnitude of a mercury problem arising from impoundment. Its use in a risk assessment should be combined with complementary information on the rate of flooding, the morphometry of the basin being flooded, and the nature of the flooded material. On the basis of what is presently known about the causes of reservoir mercury problems, it is reasonable to expect that a reservoir development which involves extensive flooding of relatively flat, well vegetated terrain over a moderately long period of time (e.g., two years), is likely to experience significant post-impoundment increases in fish mercury levels.

8.1.4 Erosion Potential

Our examination of the factors which we presumed to be associated with the potential for erosion to occur in a reservoir, and those associated with a related factor, resuspension and sedimentation, did not produce any striking results. At SIL, continued erosion of shoreline materials has been suggested as providing an ongoing supply of organic matter to stimulate bacterial methylation (Bodaly et al., 1984b). Erosion does not

appear to be a necessary factor, however, as evidenced by the elevated mercury levels in reservoirs on the La Grande River, where erosion was apparently negligible (Boucher and Schetagne, 1983).

Just as the nature of the flooded material is important in a consideration of the role of extent of flooding (8.1.3 above), the nature of the eroded material must be considered when evaluating the impact of erosion. Mesocosm studies in a polluted system (Rudd and Turner, 1983) and our microcosm studies both suggest that the presence of suspended inorganic matter may inhibit mercury accumulation in fish. Mesocosm studies at SIL, however, found additions of inorganic matter to stimulate uptake. One explanation for this difference might be that inorganic matter strips out mercury when its concentration in water is high (polluted system) but releases it when concentrations are low (unpolluted systems) (R.E. Hecky, personal communication). While this reconciles the results of the two mesocosm studies, it does not explain our microcosm results. Since it was not our objective to examine the role of suspended inorganic matter in our microcosms, further experiments would be required to properly address this question in the laboratory.

One potentially important role that erosion might play in a reservoir mercury problem is to delay the eventual decline of fish mercury levels back to background concentrations. Shoreline erosion may provide an ongoing supply of suspended materials and organic matter to the reservoir, thus stimulating increased methylation and uptake for as long as it takes for the shoreline to stabilize. The actual effect of erosion may also change through time as the nature of the eroded material changes, perhaps from predominantly organic matter as surface soil layers are removed, to clays and other inorganic materials found in deeper horizons (R.E. Hecky, personal communication).

In summary, while there are reasons to presume that erosion potential will have an important influence on post-impoundment mercury levels, the magnitude, and in fact even the direction of its effect is far from well understood. This is clearly an area where further study is needed; on the

basis of the present evidence one can only say that at sites where erosion potential is great, the possibility exists for an exaggerated and prolonged mercury problem.

8.1.5 Fish Species Present

There is little doubt that fish species occupying differing positions in the food web exhibit consistent differences in relative mercury levels, although the magnitude of these differences appears from our analysis to vary regionally (Table 25). Top predator species such as pike and walleye invariably have the highest mercury levels of all fish in a reservoir. Thus, if a proposed reservoir is unlikely to contain such species, the potential for a problem is considerably reduced. As for any problem that involves biomagnification, the shorter the food chain, the less the opportunity for the substance to reach unacceptable levels. This factor is unlikely to be of much importance, however, since most new impoundments can be expected to contain at least one piscivorous fish species. As well, these species are normally the ones of the greatest socio-economic importance.

8.1.6 Summary

For any proposed development, the five factors discussed above can be evaluated, using observations that would be taken as part of a pre-development site assessment, or on the basis of existing information from existing water bodies. Since the relative importance of these factors appears to vary regionally, it is inappropriate to assign general weightings to each factor. Obviously, the more factors which appear to favour elevated mercury, the greater the expected risk of a mercury problem arising.

To provide a final emphasis for our earlier remark that this rating scheme should not be expected to work in all circumstances, consider again the Nova Scotia reservoirs. Wreck Cove Lakes reservoir has a higher

pH than Cheticamp reservoir, the lowest relative area flooded of the four reservoirs, a smaller fetch than Cheticamp or Gisborne, a greater mean depth than Cheticamp, similar soils to the other three nearby reservoirs, and only contains brook trout. Most of all, it is in the same watershed as three other reservoirs which contain brook trout having reasonably low mercury levels. Thus the hazard rating for this reservoir would have to be quite low, yet observed post-impoundment mercury levels are unusually high. On the basis of the information presently available for Wreck Cove Lakes, even with the advantage of hindsight, we can offer no explanation for the observed high mercury levels in brook trout.

In the absence of a proven scheme for predicting the effect of impoundment on fish mercury levels, and since the prospect of a mercury problem is only one of a host of factors that influence the decision to construct a reservoir, it is very important that two other issues be examined as part of an overall assessment of the mercury problem. First the potential for mitigation against anticipated problems and rehabilitation of existing problems should be evaluated. Second, areas of research and monitoring should be identified, which offer the greatest potential for improving our ability to predict the magnitude and duration of a reservoir problem.

8.2 POTENTIAL METHODS FOR MITIGATION/REHABILITATION

At the first workshop, one of the first topics discussed concerned the set of possible actions that might be taken to mitigate against or rehabilitate a mercury problem. The following list emerged from this discussion:

1) resuspend clay-rich sediments;

2) cover bottom with inorganic materials;

3) add selenium to the reservoir;

4) add lime to the reservoir to increase pH;

5) add sulphate or sulphite salts to bind mercury;

6) add nutrients to stimulate production and achieve "biomass dilution";

7) stock fish to achieve biomass dilution;

8) regulate fish harvests;

9) encourage utilization of non-piscivorous fish;

10) process fish to remove mercury prior to consumption;

11) control erosion; and

12) remove vegetation prior to flooding.

At the second workshop, each of these options was discussed in some detail, particularly in light of the results of this and other studies. In this section we provide a summary of these discussions.

8.2.1 Resuspension of Clay-Rich Sediments

We have already discussed the conflicting evidence for the effectiveness of this control measure. While Rudd and Turner (1983) found this action to reduce uptake in a polluted system, mesocosm experiments conducted in an unpolluted reservoir (R.E.Hecky, personal communication) demonstrated the opposite result. Our first microcosm experiment provided limited evidence in support of the former result. These considerations alone cast serious doubt on the wisdom of taking this action without further studies to measure its effect.

Newly formed reservoirs often have high suspended sediment loads initially even without actions being taken to deliberately encourage this. To maintain this condition after the reservoir has begun to stabilize would probably be prohibitively expensive in most reservoirs. As well, high suspended sediment loads may have other undesirable side effects,

such as reducing aquatic productivity in already oligotrophic systems.

8.2.2 Cover Bottom with Inorganic Matter

The effectiveness of this action was also demonstrated by Rudd and Turner (1983), again for a polluted system. The inorganic layer presumably isolates the substrate for methylating bacteria from the sediment-water interface. In a relatively large system such as most reservoirs, the water mass is unlikely to be sufficiently stable throughout (and especially in the littoral zone) to allow this condition to persist. Drawdown activities will probably act to further reduce the persistence of an inorganic layer. Perhaps most important, the method will only be useful if an external source of organic matter, such as shoreline erosion, is not available.

As for the previous, related action, the scale of manipulation required to make the action work, even if conditions were suitable, would most likely make it impractical to undertake. Even if this action were practical, considerable testing at a pilot scale would be essential to determine its effectiveness. Such testing should at least ensure that undesirable side effects, such as losses of benthic production, do not arise.

8.2.3 Add Selenium to the Reservoir

Selenium appears to be effective at both reducing the uptake of mercury by biota and its toxicity to biota (Turner and Rudd, 1983). The levels of addition for which obvious positive effects are found, however, approach the levels set as water quality standards. It is unlikely that additions of a chemical substance with known toxic effects at such high levels would ever be considered publicly acceptable. In fact, health officials in Manitoba have repeatedly emphasized that selenium additions are not an acceptable solution to a mercury problem (R.E. Hecky, personal communication). In Quebec as well, this action was ruled out as a potential mitigative measure (G. Guertin, personal communication). An alterna-

tive measure which has been proposed is to provide dietary supplements of selenium for groups consuming large quantities of contaminated fish. This approach has also been ruled out by health experts in Manitoba.

8.2.4 Add Lime to Increase the pH

The idea of liming lakes to increase pH has been the focus of much debate in the acid rain literature. In that context, the ultimate objective is to increase the pH. Here, a further link is required: the increased pH must have the effect of reducing mercury levels in fish. While our analyses show that pH (and calcium) appear to be important variables, they do not offer a causal explanation for this effect. The recent work on methylation and demethylation (Ramlal et al., 1985a; Xun et al., 1985) demonstrate that the role of pH is by no means simple. Sloan and Schofield (1983) did not find that mercury levels changed after liming, even though the pH increased. This result, the fact that the action is an indirect one which makes a number of assumptions about the cause of the mercury problem, and the impracticality of its application in situations such as run-of-river reservoirs, casts serious doubt on the wisdom of using this method of mitigation.

8.2.5 Add Sulphate or Sulphite Salts to Bind Mercury

This action must be considered in much the same light as the addition of lime. It makes assumptions about the mechanisms by which mercury becomes available for accumulation in fish, and will only be effective under certain conditions (e.g., a reducing environment). Given the potential side effects of these additions, it is very unlikely that this action would ever be considered acceptable.

8.2.6 Add Nutrients

Once again, this action makes strong assumptions about the nature of the mercury problem. The literature contains conflicting views on whether

nutrient additions will enhance or suppress mercury bioavailability (D'Itri et al., 1971; Rudd and Turner, 1983). Certainly the concept of biomass dilution, which argues that more productive systems dilute the available mercury among the greater biomass present, has been severely criticized. The concept seems to assume that the supply of mercury is limiting, which is very unlikely given the very small proportion of the total mercury in a system that is in living organisms.

All of the actions discussed so far involve mitigation of an existing problem by adding something else to the reservoir to prevent or reduce the accumulation of mercury by fish. In every case, there is uncertainty about whether the treatment will even work, and a potentially long list of undesirable side effects can be imagined. Certainly nutrient additions is no exception to this. These two considerations are sufficient to argue compellingly that none of the actions so far discussed can be given serious consideration as options for mitigation, at least before more is known about the problem.

8.2.7 Stock Fish

The primary motive for stocking fish derives again from the concept of biomass dilution, by assuming that adding more biomass in the form of fish will dilute the available supply of mercury. As mentioned above, this concept has been severely criticized (see section 2.3), and does not appear to be consistent with what we know about the partitioning and dynamics of mercury in aquatic systems.

Stocking fish in a reservoir where a mercury problem already exists would also probably be very difficult to justify, without being very certain that the action would be effective. An interesting option that was suggested at the second workshop would be to stock fish in nearby water bodies where the problem does not exist, thereby compensating users for lost opportunities in one system by increasing opportunities elsewhere. Such an approach is in accordance with the recently adopted policy of 'no net loss' of the Department of Fisheries and Oceans.

8.2.8 Regulate Fish Harvests

This may be the most feasible action to take, although its suitability may depend greatly on a key question: how long will the problem last before the regulations can be relaxed? There are at least two different approaches.

First, one could simply restrict fishing on a reservoir, such that individual harvests are consistently low enough to minimize risks of excessive mercury intake by humans. Any action short of completely closing the reservoir, however, would probably be virtually impossible to police. This is apparently the action that would be taken in some jurisdictions (e.g., B.C., T.A. Watson, personal communication).

Where commercial or native food fisheries operate, this measure would almost certainly require some sort of compensation for the fishermen. In Manitoba, compensation for decline in catch and value of whitefish on a year-to-year basis was originally used for SIL, but when the problem turned out to last for several years, this became very costly and an alternative, one-time settlement was sought (R.E. Hecky, personal communication). Thus it is clear that the anticipated duration of the problem is a key piece of information, and one about which we remain very uncertain.

The second option involves regulating harvests in a manner that shifts the age structure of the fish population towards smaller, younger fish. This allows one to take advant ` of the fact that smaller fish generally have lower mercury concentrations. Such shifts in harvest age structure can be accomplished in two ways. First, exploitation rates can be increased (Gothberg, 1983), thereby allowing fewer fish to recruit to older age classes (by increasing mortality rates earlier in life). This action may be difficult to implement in remote, effort-limited areas (e.g., northern Manitoba). At the other extreme, one must be careful not to overexploit the fish population such that long term returns to the fishery are severely reduced. A second, less intense approach would be to impose minimum and maximum size limits for the catch (so called slot

limits), thereby protecting young fish from exploitation before maturing and minimizing the harvest of older fish with relatively high mercury levels.

8.2.9 Encourage Use of Non-Predatory Fish

This action would obviously have considerable merit, given the widespread observation that non-predatory fish have lower mercury concentrations (e.g., Wren et al., 1983; this study, Table 25). Aside from the obvious situation where planktivorous fish are prized by sports fishermen (e.g., brook trout, kokanee), however, it is unlikely that such encouragement would have very much impact on the choice of anglers, or other fishermen.

8.2.10 Treat Fish Prior to Consumption to Remove Mercury

This action is only practical for fish caught commercially, since it involves blending fish with relatively high mercury levels together with those containing less mercury. Even for commercially caught fish it is only practical when the fish are being processed before marketing, rather than sold as whole fillets. Unfortunately, predatory fish are marketed most often as whole, unblended fillets.

8.2.11 Control Erosion

In an earlier discussion we concluded that the role of erosion in the reservoir mercury problem was by no means clear. For reservoirs where considerable erosion is occurring, attempts at control will probably be prohibitively expensive. Further elucidation of the mechanisms by which erosion affects mercury accumulation in fish is needed before erosion control actions should be seriously considered as a mitigative measure. One action related to erosion control, the planting of vegetation in the drawdown zone, is particularly questionable, since this involves increasing the amount of organic matter in the reservoir (which may promote methylation) at the same time as controlling erosion.

8.2.12 Remove Vegetation Prior to Flooding

If, as suggested by Bodaly et al. (1984b), the inundation of large amounts of organic matter which stimulate methylation is a key factor in giving rise to a reservoir mercury problem, then clearly this action would be of value. Our statistical analysis did not show this, however, since the percent area cleared was not a good predictor of fish mercury levels. As we stated earlier, this is likely because the reservoirs we described as having been cleared had often had only merchantable timber removed (removal of merchantable timber is now standard practice in some regions such as B.C., T.A. Watson, personal communication). From the recent mesocosm results at SIL for spruce bough additions one might expect that removal of trees would only reduce the magnitude of the initial increase of mercury, but not the duration of the problem, since spruce stimulated the greatest uptake rates, but the most rapid breakdown (Crawford and Rosenberg, 1984).

Removal of more than just the merchantable timber from a reservoir may prove very costly. For larger reservoirs it may be virtually impossible to remove significant amounts of vegetation beyond the standing timber. Before even considering this action, it would be useful to estimate the proportion of the total organic matter in the area to be flooded that is contained in different compartments (e.g., trees, shrubs, mosses, soil organic layers). This would give one some idea of what would be accomplished by removing only certain types of vegetation. Even with this information, however, it would be very difficult to justify substantial expenditures on vegetation removal without a better understanding of its effectiveness.

8.2.13 Summary

It is clear from the foregoing discussion that there are no obvious mitigative or rehabilitative measures that can be taken to deal with the reservoir mercury problem. There are two principal reasons for this. First, we remain sufficiently uncertain about the factors that cause the

problem that it is hard to justify costly measures which assume a particular cause. Second, most of the actions discussed have the potential to lead to other effects which may be less desirable even than the mercury problem itself. Since different factors appear to be important for different reservoirs, recommendations for mitigative measures should invariably be preceded by investigations of these factors and/or by pilot study tests of proposed control measures at the site in question.

The only actions which presently show promise in general are those associated with the harvest of fish. If regulations can be imposed which encourage exploitation of fish with relatively low mercury levels, either by stocking nearby water bodies, or by shifting the age/size structure of the catch, then the problem can at least be avoided. Given our present level of knowledge, this is probably more desirable than taking risky, uncertain measures to solve the problem. Decisions regarding the necessity of implementing or continuing harvest controls could be based on the monitoring programs detailed below.

8.3 PRIORITIES FOR FUTURE RESEARCH

At the second workshop, it was agreed that the most important issue surrounding the mercury problem concerns its duration. In section 7.2 we pointed out that no single reservoir exists for which post-impoundment increases and subsequent declines have been clearly observed. Further, the limited evidence available suggests that the duration of the problem may be quite variable among reservoirs. The trade-off between direct compensation for lost opportunities and investment in costly, and risky, control measures, depends critically on how long the problem is expected to last.

For new developments, it is important to have baseline, or pre-impoundment data. Three to four years of pre-impoundment data will probably be necessary to characterize among-year natural variability. After impoundment, data should be collected annually for several years and then less frequently as the levels return to pre-impoundment values. Where

pre-impoundment data are not available, data from nearby lakes offer an alternative. Re-sampling of existing reservoirs (as we did for 3 Ontario reservoirs, and 1 in B.C.) would also be informative. It would probably be sufficient for the purposes of this question to only sample fish, but it is critical that a common species be used for comparison (at least within regions).

The existing database developed in this study could be considerably improved with relatively little effort. Physical and chemical data are missing for most reservoirs, and many of these parameters could be easily obtained from a simple field sampling program. As well, measures that are used presently to represent certain factors could probably be greatly improved in some cases, especially for vegetation, drawdown and temperature. For vegetation more detailed information may be available from forestry and agricultural maps. Information on the frequency and timing of drawdowns may be more useful than the data on magnitude that is currently being used. For temperature, measurements from within the reservoir would obviously be more valuable (but harder to obtain) than air temperature degree-days. Other measures could also be added, such as rate of flooding, shoreline development and observed methylation rates (method of Ramlal et al., (1985)). This latter measure could provide a comparable measure of the impoundment effect for all reservoirs, thus avoiding the problem of fish species differences. For this to be justified, however, a good relationship between methylation rates and mercury levels in fish would have to be established.

The database could also be improved by adding data from new reservoirs as more fish mercury data becomes available, and by including data from reference lakes. These latter data could probably be obtained in part from existing sources such as the NIS database for eastern Canada, discussed earlier. With such information, and with more complete and representative physical and chemical data, the value of the database would be vastly improved.

Experimental studies of the factors presumed to influence reservoir fish

mercury levels also remain a research priority. Microcosms should not be ruled out as experimental tools, but they should be used cautiously, and the duration of the experiments should be longer than used in this study. Some questions which might be explicitly addressed using microcosms concern the effect of suspended inorganic matter and of covering the sediments with "clean" materials. Also the effect of changing water quality (e.g., pH, hardness) could be examined. Mesocosms offer a desirable alternative to microcosms for addressing any of these questions and are more realistic. In making the choice between micro- and meso- one has to consider the difficult trade-off between cost (and thus replicability) and realism.

A third experimental approach that warrants consideration is to place uncontaminated fish in cages in reservoirs and observe the change in their mercury levels. By placing such cages in different reservoirs or in different parts of the same reservoir (with differing physical/chemical conditions) one could empirically assess the role of certain factors without deliberately manipulating the system. This approach would obviously require that the fish be fed. The advantage of using caged fish rather than simply collecting fish from the wild is that fish in the wild (especially large piscivores) will tend to integrate over large spatial areas, thereby potentially averaging the effects of a wide range of conditions.

Finally, we would recommend that a pilot scale study of increased harvesting be conducted, perhaps in a sub-basin of a large reservoir, to see whether this potential control measure is effective. Such a study should have a control, which would ideally be another sub-basin of the same reservoir. Harvesting could be encouraged in one basin, and restricted in another. If monitoring indicated that fish movements between basins was minimal, or if movement was restricted in some way, this approach should be effective.

8.4 UTILITY OF THE AEAM APPROACH

When the methodology known as Adaptive Environmental Assessment and Management was originally developed (Holling, 1978), a central focus was the development of computer simulation models of ecological systems. In our proposal to carry out the work described in this report, we had envisaged developing such a model, to describe the processes by which mercury accumulation apparently increases in fish when new reservoirs are created. At the initial workshop for this project, however, we soon came to recognize that this was not a realistic objective, given the great complexity of this issue, and the uncertainty associated with many of the key processes of interest. Thus it might be imagined that this is an example of a problem where the AEAM approach was not applicable.

AEAM is much more than just building simulation models, however. In fact, in recent years, more and more applications of the approach have not even included the development of a simulation model as an objective. It is our view that simulation modelling is but one of many tools of AEAM, and that the essential ingredients of the methodology are carefully planned research activities (and management decisions) that are based on a consensus among both technical experts and decision makers. In other words, the approach strives to ensure both scientific rigor and policy relevance, at the same time. The key to achieving this goal is communication, both among scientists and between scientists and decision makers. This communication is achieved principally through the use of interdisciplinary workshops. For this project, the most important activity that took place was the first workshop, described earlier in this report (section 3) and in more detail elsewhere (Marmorek et al., 1984). It was at this workshop that the research plan for the project was formally discussed. Although it could be argued that the major activities of the project did not change as a result of the workshop (all three were described in our original proposal), the nature and relative importance attached to each was greatly influenced by the workshop discussions. For example, in our proposal we envisaged the microcosm experiments as addressing very detailed cause-effect questions which would help us to

understand some of the individual processes affecting bacterial methylation. Discussions at the workshop made it clear that microcosms, if used at all, were better employed in examining more empirical questions such as the role of differing types of vegetation on mercury accumulation in biota.

The first workshop was therefore extremely beneficial to the members of the project team, both as a learning experience and as an opportunity to determine the needs and interests of the various utilities. Perhaps more important, the workshop was viewed by most of the invited participants as a valuable exercise, particularly in terms of the sharing of ideas and experiences from different regions of the country. Finally, the workshop was largely responsible for ensuring that the project had an interregional emphasis, thus making it unique amongst studies of the reservoir mercury problem.

The second workshop was also very successful, especially in providing an opportunity for the project team to discuss their results with technical experts. As well we received valuable advice on the content of our final report, particularly regarding the subjects discussed earlier in this section. In our view, this information could not have been shared between the client and the project team nearly as effectively without the workshop.

In conclusion, we would argue that this project has derived considerable benefit from the AEAM methodology, through the use of interdisciplinary workshops. We believe strongly that the nature of the study would have been quite different (and the results less useful) if the workshops had not been used to guide our activities. Thus we would not hesitate to recommend the use of this approach for future projects of this type.

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APPENDIX A

LABORATORY ANALYTICAL METHODS FOR

WATER, SEDIMENT, AND FISH TISSUE SAMPLES

1 PREPARATION

Water samples required no preparation beyond the preservation at the time of sampling as described in Section 5.1.1.

Sediment samples from the reservoirs were allowed to thaw to room temperature. For mercury analysis, sub-samples of approximately 0.25 g (wet weight) were weighed on a Mettler (Model HK 160) balance directly into acid washed Teflon bombs. A mixture of concentrated sulphuric and nitric acids (4 H_2SO_4 : 1 HNO_3) was added to the Teflon bombs containing sediment. The mixture was sealed with a Teflon screw cap and placed in a pressure cooker. The digests were heated to establish pressure within the cooker, and this condition was maintained for 6 to 8 hours. The digests were then allowed to cool in the bombs until they attained room temperature. The digested sediment was poured directly into 25 ml volumetric flasks. The bombs and lids were then rinsed at least 3 times with distilled water, and this rinse was poured into the volumetric flasks until a volume of 25 ml was reached. An NBS (National Bureau of Standards) sample was also subjected to the same analytical procedure.

Fish samples collected in the field and those generated through the microcosm experiments were treated identically. The first step in the digestion of fish tissue samples was to homogenize either a 5 g section of tissue (field samples), or the entire fish (microcosms) in a Sorvall blender. Samples were then placed in capped plastic Scintillation vials and frozen until further analysis could take place. When required, frozen samples were thawed, and a piece of tissue of 0.25 g weight was placed in a 30 ml Teflon digestion bomb. 2.5 ml of a 4:1 mixture of H_2SO_4 : HNO_3 was added to the Teflon bomb, which was sealed and placed in a cooker at 80 ^{0}C for 2 hours, or until the digest was a clear, pale yellow

liquor. Following this, the digest was decanted into a glass container, and diluted to 25 ml with distilled water.

2 ANALYSIS

2.1 Mercury Determination

The cold vapor atomic absorption technique using an LDC mercury monitor 1235 was employed to determine mercury levels in all water, sediment, and fish tissue samples. 1% $NaBH$ in 0.05 M KOH was used as the reagent for water and sediment samples; 10 g of $SnCl$ + 5 g $(NH_2OH)_2H_2SO_4$ + 2.5 g NaCl in 250 ml of 10% HCl was used for analysis of fish tissue.

A Buchler Polystatic peristaltic pump set at 5 ml/capillary tube/min was used with two feeder lines of 1.5 mm internal diameter Tygon capillary tubing. These feeder lines were connected to a 3.0 mm internal diameter mixing coil with a Tygon Y-junction. The mixing coil joined to a 3.0 mm internal diameter air line with a glass Y-junction immediately prior to a liquid-gas separator. The gas phase line was connected to a drying tube packed with layers of cotton and silica gel and a 30 cm mercury lamp in the LDC mercury monitor. The gas phase line from the monitor connected to a charcoal gas scrubber filled with activated charcoal in distilled water, which was vented into a fume hood. A Gilmont flow meter was connected in-line between the mercury monitor and charcoal scrubber. Air flow rate was set at 2500 ml/min for fish and sediment, and at 1500 ml/min for water. The mercury monitor range switch was placed at 0.02 for fish and sediment, and at 0.01 for water. The Omniscribe chart recorder input was set at 10 mV.

The procedure for analysis was as follows: one capillary tube was placed into the reductant; the other was used to aspirate the sample for 15 seconds. The sample capillary was then rinsed for 60 seconds, and the peak height recorded. A calibration curve for the samples was produced by analyzing solutions with known concentrations of mercury. National Bureau of Standards (NBS) samples and blanks were run with each set of

samples; NBS readings were always within 5% of expected values.

2.2 Loss on Ignition

A measure of the organic content of sediments was made by determining the % of loss on ignition (LOI). Sediment samples were weighed when wet (4-11 g) using a Mettler HK 150 balance, then placed for 3 days in a drying oven at 72-74 ^{0}C to remove all moisture. The dried samples were transferred to a dessicator, then weighed to determine dry weight. The samples were then put in a muffle furnace at 550^{0}C for 90 minutes. After transfer to a dessicator, they were weighed a final time to determine the ash free dry weight. % loss on ignition was calculated by:

$$\frac{\text{(Dry weight - ash free dry weight)}}{\text{Dry weight}} \times 100\%$$

APPENDIX B

REGRESSIONS OF LENGTH ON MERCURY CONCENTRATION FOR FISH BY SPECIES, RESERVOIR, AND YEAR

Species	Reservoir	Year	n	Regression	r^2	Signif.	Length Corrected [Hg] ug/g
Brook Trout	Smallwood	1977	89	y=.01111x-.10997	.02905	.1103	0.47*
	Hinds Lake	1980	19	y=.01320x-.17708	.66935	.0000	0.14
		1981	4	y=-.00739x+.30938	.37569	.3871	0.12**
		1982	24	y=.04246x-.85118	.64023	.0000	0.17
		1983	26	y=.02948x-.55110	.40415	.0005	0.16
	Cold Spring	1982	3	y=.01220x-.15141	.93166	.1684	0.14
		1983	40	y=.01439x-.21435	.28189	.0004	0.14
	Great Burnt	1982	3	y=.00851x+.15470	.96711	.1161	0.36
		1983	26	y=.02443x-.26537	.25841	.0080	0.32
	Cheticamp	1984	10	y=.03602x-.45413	.12588	.3144	0.41
	Gisborne	1984	10	y=.00908x+.02725	.14575	.2763	0.25
	McMillan	1984	10	y=.03184x-.44592	.70027	.0025	0.32
	Wreck Cove	1984	10	y=-.19883x+5.3590	.39657	.0510	0.94**
	Long	1984	4	y=.03210x-.54428	.55734	.2534	0.23
Lake Trout	Smallwood	1977	353	y=.01714x+.11617	.10399	.0000	1.06
	Ossokmanuan	1977	35	y=.02505x+.32305	.08650	.0864	1.70
	Long	1984	10	y=.01737x-.40286	.38695	.0548	0.55
	Caniapiscau	1978	106	y=.02317x-.70540	.39295	.0000	0.57
Atlantic Salmon	Smallwood	1977	5	y=.03728x-.70172	.89557	.0148	0.42
	Ossokmanuan	1977	3	y=.00965x-.16308	.99978	.0094	0.13
	Hinds Lake	1980	19	y=.01875x-.21110	.65459	.0000	0.35
		1981	22	y=.02072x-.05766	.63986	.0000	0.56
		1982	16	y=.04436x-.82971	.91762	.0000	0.50
		1983	52	y=.04821x-.83963	.63973	.0000	0.61
	Cold Spring	1982	35	y=.01278x+.09111	.21236	.0053	0.47
		1983	43	y=.02843x-.39753	.64810	.0000	0.46
	Great Burnt	1982	32	y=.03297x-.46081	.46608	.0000	0.53
		1983	40	y=.03140x-.35720	.67772	.0000	0.58

* actual mean mercury value used because these brook trout from this reservoir were much larger than brook trout from all other reservoirs.

** actual mean mercury value used because of negative slope

Species	Reservoir	Year	n	Regression	r^2	Signif.	Length Corrected [Hg] ug/g
Lake Whitefish	Smallwood	1977	678	y=-.00039x+.39726	.00064	.5104	0.38**
	Ossojmanuan	1977	187	y=.00537x+.08283	.24281	.0000	0.30
	Outardes 2	1984	25	y=.02737x-.59323	.39543	.0008	0.50
	Manic 5	1983	7	y=.00473x-.00107	.79636	.0069	0.19
	Opinaca	1978	25	y=.00429x-.09697	.46685	.0002	0.07
		1981	29	y=-.00073x+.29619	.00140	.8473	0.26**
	La Grande 2	1978	53	y=.00485x-.08677	.30559	.0000	0.11
		1981	30	y=-.0311x+.61304	.04980	.2359	0.49**
		1982	90	y=.00449x+.33584	.03903	.0620	0.52
	La Grande 3	1982	30	y=.01427x+.09620	.27809	.0027	0.67
	Caniapiscau	1978	137	y=.00089x+.12879	.00564	.3832	0.16
	SIL	1975	25	y=.00070x+.04226	.05445	.2616	0.07
		1978	25	y=.00679x+.00454	.08305	.1624	0.28
		1979	25	y=.00238x+.13114	.00320	.7883	0.23
		1980	25	y=.00076x+.11520	.00212	.8269	0.15
		1981	25	y=.00184x+.07484	.00981	.6377	0.15
		1982	25	y=.00093x+.07236	.00384	.7686	0.11
		1983	25	y=.00499x-.06113	.06512	.2183	0.14
	Notigi	1980	6	y=-.02127x+1.0185	.32657	.2361	0.12**
		1981	25	y=.00810x-.01566	.54570	.0000	0.31
		1983	25	y=.00477x+.03882	.33889	.0023	0.23
	Playgreen	1981	25	y=.00240x-.05933	.10489	.1143	0.04
	Williston	1980	52	y=.00914x-.17123	.10798	.0174	0.19
Mountain Whitefish	Lower Arrow	1980	4	y=.00244x-.01263	.18447	.5705	0.08
		1984	5	y=.00280x+.07424	.33345	.3080	0.19
	Libby	1984	3	y=.00255x+.15156	.75543	.3293	0.25
Northern Pike	Smallwood	1977	175	y=.01221x+.42934	.11486	.0000	1.19
	Ossokmanuan	1977	184	y=.02766x-.83083	.52029	.0000	0.90
	Outardes 2	1984	14	y=.02000x-.68018	.40004	.0152	0.57
	Manic 1	1984	15	y=.04618x-1.7675	.17530	.1204	1.12
	Manic 5	1983	6	y=.02321x-1.0765	.48998	.1215	0.23
	Opinaca	1978	16	y=.00788x-.26162	.15890	.1262	0.23
	La Grande 2	1978	50	y=.01405x-.40087	.37278	.0000	0.48
		1982	90	y=.00979x-.69927	.11354	.0012	1.31

Species	Reservoir	Year	n	Regression	r^2	Signif.	Length Corrected [Hg] ug/g
Northern Pike, continued	La Grande 3	1982	30	y=.00415x-.69938	.04256	.2740	0.96
	Caniapiscau	1978	57	y=.02212x-1.1408	.34854	.0000	0.24
	Aubrey	1977	15	y=.02187x-.33481	.33021	.0250	1.03
		1984	11	y=.01412x-.23695	.64596	.0029	0.64
	Tunnel	1977	15	y=.04310x-1.6872	.84078	.0000	1.01
		1984	12	y=.00977x-.13456	.35315	.0416	0.48
	Red Rock	1977	15	y=.02775x-.87480	.89265	.0000	0.86
		1984	8	y=.03240x-1.1362	.45945	.0647	0.89
	Dog	1978	25	y=.01975x-.46275	.75277	.0000	0.77
	SIL	1978	25	y=-.00237x+.76357	.01638	.5420	0.62**
		1979	25	y=.01438x-.10988	.40263	.0007	0.79
		1980	25	y=-.01341x+1.4365	.16663	.0428	0.64**
		1981	25	y=.02022x-.49794	.53407	.0000	0.77
		1982	25	y=.03387x-1.2599	.37937	.0010	0.86
		1983	25	y=.01908x-.35381	.23024	.0152	0.84
		1984	25	y=.03580x-1.6174	.33109	.0026	0.62
	Notigi	1980	5	y=-.01938x+3.4750	.07479	.6562	1.95**
		1981	25	y=.04580x-1.0239	.46863	.0002	1.84
		1982	10	y=-.01532x+2.9783	.02778	.6453	1.85**
		1983	25	y=.03028x-.46168	.79982	.0000	1.43
	Playgreen	1978	5	y=.00427x-.03785	.17701	.4806	0.23
		1981	25	y=.00758x-.17403	.24082	.0127	0.30
	Cedar	1970	23	y=.01656x-.28529	.65974	.0000	0.75
		1971	25	y=.01899x-.54060	.39399	.0008	0.65
		1972	25	y=.02690x-1.0368	.57982	.0000	0.77
		1973	25	y=.02517x-.97185	.76663	.0000	0.60
		1976	6	y=.06514x-3.2084	.90204	.0037	0.86
		1977	20	y=.00796x-.22202	.09478	.1867	0.28
		1978	25	y=.01098x-.36963	.32645	.0029	0.32
		1979	6	y=.00719x-.11901	.14586	.4550	0.33
		1980	5	y=.01472x-.63469	.22942	.4143	0.29
		1981	5	y=.01851x-.72029	.51513	.1722	0.44
	Brazeau	1984	17	y=.00086x+.26974	.02234	.5669	0.32

Species	Reservoir	Year	n	Regression	r^2	Signif.	Length Corrected ^{t}Hg ug/g
Walleye	Opinaca	1978	25	y=.02033x-.47733	.50124	.0001	0.44
		1981	30	y=.04218x-.68442	.62887	.0000	1.22
	La Grande 2	1978	53	y=.05717x-1.5053	.61245	.0000	1.07
		1981	29	y=.02925x-.04332	.35254	.0007	1.27
		1982	90	y=.03520x+.63406	.20600	.0000	2.22
	La Grande 3	1982	30	y=.00331x+1.6245	.01215	.5620	1.77
	Aubrey	1984	19	y=-.00127x+.48448	.00287	.8276	0.44**
	Tunnel	1977	14	y=.03819x-.48407	.84005	.0000	1.24
		1984	21	y=.02156x-.29013	.52973	.0002	0.68
	Red Rock	1977	9	y=.03414x-.43998	.82948	.0006	1.09
	SIL	1978	25	y=-.02403x-1.7228	.10738	.1098	0.65**
		1979	25	y=.01679x-.12056	.40345	.0006	0.63
		1980	25	y=.00254x+.37134	.02059	.4938	0.49
		1981	25	y=.02817x-.52557	.14546	.0600	0.74
		1982	25	y=.02380x-.36042	.20413	.0234	0.71
		1983	25	y=.02752x-.58822	.34653	.0020	0.65
		1984	6	y=.08125x-3.2242	.88992	.0047	0.43
Notigi		1978	19	y=.07556x-1.7712	.05637	.3277	1.63
		1980	4	y=.14785x-3.7925	.35062	.4079	2.86
		1981	25	y=.03601x+.36907	.30836	.0040	1.99
		1982	6	y=.04982x-1.2785	.68478	.0421	0.96
		1983	25	y=.06005x-.88530	.72444	.0000	1.82
Playgreen		1978	10	y=.01858x-.59001	.30305	.0991	0.25
		1981	25	y=.02261x-.67966	.45314	.0002	0.34
Cedar		1971	25	y=.01933x-.30074	.43860	.0003	0.57
		1973	25	y=.02979x-.85896	.25053	.0108	0.48
		1975	6	y=.05540x-2.2404	.52793	.1019	0.25
		1976	6	y=.03698x-1.2426	.61180	.0660	0.42
		1977	11	y=.01060x-.22175	.61327	.0044	0.26
		1978	25	y=.01444x-.40357	.73769	.0000	0.25
		1979	7	y=.00446x+.01999	.10834	.4710	0.22
		1980	5	y=.00563x-.00274	.19442	.4573	
		1981	6	y=.01181x-.31471	.19226	.3844	

Species	Reservoir	Year	n	Regression	r^2	Signif.	Length Corrected [Hg] ug/g
Walleye, continued	Cookson	1979	14	y=.02186x+.28090	.43106	.0108	1.26
		1980	58	y=.02939x-.34201	.69661	.0000	0.98
		1983	27	y=.02217x-.43151	.74725	.0000	0.57
White Sucker	Smallwood	1977	61	y=.00933x-.05596	.10662	.0102	0.33
	Long	1984	10	y=.00150x+.19641	.00490	.8477	0.26
	Outardes 2	1984	68	y=.00563x+.27516	.00899	.4420	0.51
	Manic 1	1984	30	y=.03261x-1.0861	.38590	.0002	0.25
	Aubrey	1984	10	y=.03061x-1.0314	.56895	.0117	0.22
	Tunnel	1984	10	y=.00346x+.02382	.17961	.2223	0.17
	Red Rock	1984	10	y=.00648x-.05230	.38548	.0554	0.21
	Playgreen	1981	25	y=.00458x-.12195	.41059	.0006	0.08
	Cookson	1980	30	y=.01288x-.15716	.08660	.1144	0.37
		1983	20	y=.00058x+.26462	.00031	.9414	0.29
	Brazeau	1984	10	y=.01835x-.50322	.76782	.0009	0.25
	Williston	1980	5	y=-.02750x+1.4025	.24395	.3977	0.31**
Longnose Sucker	Smallwood	1977	162	y=-.00296x+.53190	.04212	.0088	0.42**
	Ossokmanuan	1977	24	y=.00669x+.02763	.63457	.0000	0.30
	Opinaca	1978	8	y=.00160x+.04519	.07162	.5217	0.11
	La Grande 2	1978	50	y=.00654x-.13044	.25209	.0002	0.14
		1982	90	y=.00817x+.09333	.12519	.0006	0.24
	La Grande 3	1978	30	y=.01176x-.04562	.24579	.0053	0.44
	Caniapiscau	1978	114	y=.00937x-.23797	.40422	.0000	0.15
	Arrow	1984	7	y=.01184x-.17365	.45041	.0988	0.31
	SIL	1979	25	y=.00145x+.04868	.06099	.2340	0.11
		1981	25	y=.00724x-.15956	.08292	.1628	0.14
		1982	25	y=.00902x-.27081	.31831	.0033	0.10
		1983	25	y=.00330x-.04937	.08325	.1619	0.09
Largescale Sucker	Arrow	1980	4	y=-.08336x+3.6309	.68839	.1703	0.19**
		1984	10	y=.00163x+.11894	.29278	.1063	0.19
	Libby	1984	10	y=.00842x-.11185	.11860	.3298	0.23

APPENDIX C

POST-IMPOUNDMENT DATA FOR RESERVOIRS INCLUDED IN FISH MERCURY DATABASE

RESERVOIR	% area flooded by soil type					% area flooded by vegetation type			
	ROCK	SAND	CLAY	SILT	OTHER	CONIFEROUS STAND	DECIDUOUS STAND	OPEN VEGETATION	CLEARED
Smallwood	.10	.70	.00	.00	.20	.80	.00	.20	.00
Ossokmanuan	.10	.70	.00	.00	.20	.80	.00	.20	.00
Hinds Lake	.00	.80	.00	.00	.20	.00	.00	.20	.80
Cold Spring	.05	.50	.00	.00	.45	.00	.51	.49	.00
Great Burnt	.05	.50	.00	.00	.45	.00	.21	.79	.00
Cheticamp	.00	.70	.20	.00	.10	.00	.00	.80	.20
Gisborne	.00	.70	.20	.00	.10	.00	.00	.00	1.00
McMillan	.00	.70	.20	.00	.10	.00	.00	.15	.85
Wreck Cove	.00	.70	.20	.00	.10	.00	.00	.20	.80
Long	.00	.70	.30	.00	.00	.00	.00	.00	1.00
Outardes 2	---	---	---	---	---	.70	.10	.00	.20
Manic 1	---	---	---	---	---	.75	.15	.00	.10
Manic 5	---	---	---	---	---	.75	.05	.05	.15
Opinaca	.00	.13	.56	.31	.00	.81	.00	.08	.11
La Grande 2	.00	.13	.56	.31	.00	.94	.00	.00	.06
La Grande 3	.30	.20	.35	.15	.00	1.00	.00	.00	.00
Caniapiscau	---	---	---	---	---	.77	.00	.10	.13
Aubrey	.00	.55	.00	.45	.00	---	---	---	---
Tunnel	.00	.55	.05	.40	.00	---	---	---	---
Red Rock	.50	.30	.00	.20	.00	---	---	---	---
Dog	.05	.50	.00	.45	.00	---	---	---	---
SIL *	.20	.10	.45	.25	.00	.10	.60	.25	.05
Notigi	.20	.10	.45	.25	.00	.10	.60	.25	.05
Playgreen	.10	.10	.50	.30	.00	.60	.20	.20	.00
Cedar	.00	.50	.20	.30	.00	.60	.30	.10	.00
Cookson	.00	.15	.00	.85	.00	.00	.00	1.00	.00
Brazeau	.00	.60	.10	.10	.20	.20	.30	.50	.00
Williston	.05	.50	.20	.20	.05	.25	.70	.00	.05
Arrow	.15	.10	.50	.25	.00	.90	.10	.00	.00
Libby	.10	.50	.20	.20	.00	.00	.00	.00	1.00

* For the analyses presented in Section 7, SIL was assumed to have a soil type of 88% rock and 4% sand, clay and silt. These assumptions were subsequently shown to be invalid (R.E. Hecky, personal communication) and the values for Notigi reservoir are used in this table as a more realistic approximation to the true values. In our view, this error does not influence the conclusions presented in Sections 7 and 8.

RESERVOIR	FETCH km	VOLUME $10^6 m^3$	MEAN DEPTH m	MAX. DEPTH m	REL AREA FLOODED	AVG. DRAWDOWN m	MAX. DRAWDOWN m	DRAINAGE AREA ha	RUNOFF mm/yr
Smallwood	83.2	28970.0	4.7	---	.500	3.65	8.69	4517000.	600.
Ossokmanuan	25.7	2850.0	---	---	.250	3.66	4.11	2238800.	600.
Hinds Lake	12.0	306.0	---	38.5	.356	1.75	4.70	65110.	950.
Cold Spring	7.8	130.0	7.6	36.0	.412	0.50	1.00	----	1050.
Great Burnt	11.0	557.0	10.1	29.0	.455	0.30	0.50	1347.	1050.
Cheticamp	5.5	616.0	5.9	19.5	.958	---	6.70	4921.	1300.
Gisborne	4.5	479.1	8.1	24.0	.913	---	13.72	20070.	1300.
McMillan	2.4	50.3	10.9	48.0	1.000	---	16.50	8200.	1300.
Wreck Cove	3.0	26.3	7.4	19.2	.673	---	7.80	21787.	1300.
Long	7.8	48.9	7.3	27.4	.250	4.00	5.30	7770.	650.
Outardes 2	---	---	---	---	----	---	0.31	1890000.	650.
Manic 1	14.2	39.0	---	---	----	1.22	1.22	4571000.	650.
Manic 5	---	141600.0	---	365.8	.808	10.70	19.90	2924000.	650.
Opinaca	---	8500.0	8.2	42.0	.712	---	4.00	4027500.	450.
La Grande 2	190.0	62400.0	22.0	100.0	.927	---	8.70	3336200.	450.
La Grande 3	---	60100.0	24.4	---	.884	---	12.20	2802400.	450.
Caniapiscau	---	72300.0	16.8	---	.798	---	12.90	3688200.	550.
Aubrey	8.0	1924.0	7.9	25.9	.788	1.65	2.89	398080.	350.
Tunnel	7.5	191.0	9.7	51.8	.897	1.00	5.52	676250.	350.
Red Rock	4.0	617.0	11.5	26.0	.709	1.00	1.60	891220.	350.
Dog	22.9	---	29.6	117.0	----	1.79	3.90	344470.	250.
SIL	45.0	23380.0	9.8	---	.157	1.00	1.30	1400000.	150.
Notigi	17.0	4100.0	7.1	29.6	.689	1.50	3.00	----	150.
Playgreen	8.7	---	4.2	12.8	.000	0.50	1.20	----	150.
Cedar	109.4	9643.0	---	30.5	.860	3.20	5.10	----	150.
Cookson	7.0	41.2	5.6	16.8	1.000	0.50	1.00	----	13.
Brazeau	9.7	711.7	---	---	.943	15.20	20.70	----	350.
Williston	165.0	76700.0	43.4	168.0	1.000	16.80	32.10	6628700.	350.
Arrow	30.9	9250.0	17.9	95.0	.200	21.00	23.20	3650000.	750.
Libby	13.5	7160.0	52.0	107.0	----	30.00	52.40	2327100.	750.

RESERVOIR	SECCHI DEPTH m	TDS mg/L	TSS mg/L	CA mg/L	pH	TOT P mg/L	CHL-A mg/L	SE mg/L	TEMP TOTAL days	TEMP GROWING days
Smallwood	4.67	11.43	---	1.32	6.57	.004	----	---	1484.4	760.2
Ossokmanuan	4.00	14.99	---	2.22	6.30	.004	----	---	1484.4	760.2
Hinds Lake	----	-----	---	----	----	----	----	---	1997.8	1070.8
Cold Spring	----	-----	---	----	----	----	----	---	2049.9	1098.2
Great Burnt	----	-----	---	----	----	----	----	---	2049.9	1098.2
Cheticamp	----	30.00	3.5	----	4.50	.013	.016	.050	2790.6	1647.9
Gisborne	----	25.00	2.0	----	6.60	.023	.002	.050	2790.6	1647.9
McMillan	----	27.00	1.8	----	6.30	.000	.001	.005	2790.6	1647.9
Wreck Cove	----	24.00	1.8	----	6.00	.019	.001	.005	2790.6	1647.9
Long	----	17.50	11.5	2.80	6.80	.030	.002	----	2494.0	1470.7
Outardes 2	----	14.65	---	----	5.45	----	----	----	2155.2	1198.9
Manic 1	----	13.99	---	----	5.30	----	----	----	2050.9	1113.7
Manic 5	6.00	13.65	---	2.40	6.00	.050	----	----	1654.1	854.6
Opinaca	2.30	8.66	---	----	6.20	.015	.002	----	1707.2	885.6
La Grande 2	1.60	16.32	---	----	6.50	.015	.002	----	1702.2	885.6
La Grande 3	----	-----	---	----	----	----	----	----	1702.2	885.6
Caniapiscau	----	-----	---	----	----	----	----	----	1445.8	732.2
Aubrey	3.00	45.20	---	----	7.11	----	----	----	2501.4	1481.6
Tunnel	3.66	39.70	---	----	6.90	----	----	----	2501.4	1481.6
Red Rock	2.60	40.60	---	----	7.50	----	----	----	2501.4	1481.6
Dog	2.50	45.10	---	----	7.50	----	----	----	2489.3	1500.5
SIL	1.19	68.50	7.7	11.00	7.76	.020	.004	----	1691.8	952.4
Notigi	2.50	114.40	1.7	8.00	7.61	----	.004	----	1770.2	999.7
Playgreen	----	198.30	23.8	35.80	8.50	----	----	----	1997.6	1174.8
Cedar	----	245.30	5.0	45.50	8.21	----	----	----	2351.7	1461.4
Cookson	----	986.80	20.7	36.00	8.50	.203	.019	.001	2741.9	1481.6
Brazeau	----	176.00	---	43.00	8.26	----	----	----	2292.3	1284.7
Williston	4.50	98.00	---	21.30	7.40	.016	.003	----	2060.4	1110.4
Arrow	7.00	76.00	1.5	15.30	8.20	.004	.001	----	2995.2	1749.0
Libby	2.40	138.00	---	----	8.20	.006	.002	----	2993.1	1758.8

Sources of the Data Presented in Appendix C:
The Number(s) Associated with each Reservoir/Data Type Entry
Correspond to the Numbered List of Sources that Follow

Reservoir	% Soil Type Information	% Veg. Type Information	Physical Information*	Water Chemistry*	Fish Community	Degree Days	Runoff	Sediment Mercury	Fish Mercury
Smallwood	38	38	10,38	15,16	1	18	20	21	11
Ossokmanuan	38	38	10,38	15,16	11	18	20	21	11
Hinds Lake	38	38	28,38	28	51	18	20	38	51
Cold Spring	38	1	1,38	1	51	18	20	--	51
Great Burnt	38	1	1,38	1	51	18	20	--	51
Cheticamp	29	56	25,33	25,56	56	18	20	60	60
Gisborne	29	56	25,33	25,56	56	18	20	60	60
McMillan	29	56	25,33	25,56	56	18	20	60	60
Wreck Cove	29	56	25,33	25,56	56	18	20	60	60
Long	12	39	19,24,50	19,24	39	18	20	24,60	60
Outardes 2	--	32	45	45	45	18	20	--	45
Manic 1	--	32	13,45	45	45	18	20	--	45
Manic 3	--	32	13,17,45	45	45	18	20	--	45
Manic 5	--	32	13,17,45	45	45	18	20	--	45
Opinaca	14	32,45	45	7,45	9	18	20	--	9
La Grande 2	14	32,45	17,45	7,45	9	18	20	--	9
La Grande 3	14	32,45	17,45	7,45	9	18	20	--	9
Caniapiscau	--	32,45	17,45	7,45	9	18	20	--	9
Aubrey	30	--	41,49,52,53	57	52	18	20	60	37,48,60
Tunnel	30	--	41,49,52,53	57	52	18	20	60	37,48,60
Red Rock	30	--	41,49,52,53	57	52	18	20	60	37,48,60
Dog	30	--	41,49,53,57	57	57	18	20	--	48
SIL	27	4	23,59	8,22,23 31,40	47	18	20	7A	47
Notigi	14	4	59	8,40,46	47	18	20	--	47
Playgreen	14	4	59	40	47	18	20	--	47
Cedar	14	4	59	40	47	18	20	--	47
Cookson	55	55	35,55	34,35	55	18	20	--	35,55
Brazeau	2	3	17,36	36	60	18	20	60	60
Williston	14	26	44,54,58	44,54,58	43	18	20	--	43
Arrow	14	26	5,44,54,58	44,54,58	60	18	20	60	42,60
Libby	14	58	44,54,58	44,54,58	60	18	20	60	60

* Physical data
- surface area
- volume
- mean depth
- max. depth
- drawdown
- fetch**
- year impounded
- watershed size

* Water chemistry data
- turbidity
- secchi depth
- TDS
- TSS
- Ca
- pH
- total phosphorous
- chlorophyll a
- Se

** often measured on maps provided by employees of the utilities

Sources of the data in Appendix C

1 Airphoto Analysis Consultants Ltd. and Beak Consultants Ltd., 1976

2 Alberta Department of Highways and Transport, no date

3 Alberta Department of Lands and Forests, 1962

4 Anonymous, 1913

5 Ash et al., 1982

6 Baxter and Glaude, 1980

7 Bobee et al., 1976

7a Bodaly and Hecky, 1979

8 Bodaly et al., 1984a

9 Boucher and Schetagne, 1983

10 Bruce, 1984

11 Bruce et al., 1979

12 Canada Department of Energy, Mines, and Resources, 1972

13 Canadian National Committee of the International Commission on Large Dams, 1976

14 Clayton et al., 1977

15 Duthie and Ostrofsky, 1974

16 Duthie and Ostrofsky, 1975

17 Efford, 1975

18 Environment Canada, 1982

19 Environment New Brunswick, 1980

20 Fisheries and Environment Canada, 1978

21 Goff, 1980

22 Hecky, 1984

23 Hecky et al., 1984

24 Hooper, 1976

25 Kelly et al., 1980

26 Krajina, 1969

27 Newbury et al., 1984

28 Newfoundland and Labrador Hydro, 1978

29 Nova Scotia Department of Agriculture, 1972

30 Ontario Department of Lands and Forests, 1965

31 Planas and Hecky, 1984

32 Quebec Ministry of Lands and Forests, 1973

Sources of the data in Appendix C (cont'd.)

33 Scheifer and Eedy, 1977

34 Surface Water Quality Committee of the International Poplar River Water Quality Board, 1979

35 Waite et al., 1980

In addition, unpublished data was made available to us through personal communication with the following individuals:

36	S. Adams	Trans Alta Utilities
37	J. Allin	Ontario Ministry of Natural Resources
38	J. Barnes	Newfoundland and Labrador Hydro
39	W. Borlund	New Brunswick Electric Power Commission
40	D.J. Brown	Manitoba Environment and Workplace, Safety and Health
41	W. Carey	Ontario Hydro
42	M.J.R. Clark	British Columbia Ministry of Environment
43	R.E. Drew	Department of Fisheries and Oceans, Vancouver
44	C. Guernsey	British Columbia Hydro
45	G. Guertin	Hydro Quebec
46	R.E. Hecky	Department of Fisheries and Oceans, Winnipeg
47	M. Hendzel	Department of Fisheries and Oceans, Winnipeg
48	A. Johnson	Ontario Ministry of Environment
49	J. Maher	Ontario Hydro
50	B. McLaughlin	New Brunswick Electric Power Commission
51	M.C. Mercer	Department of Fisheries and Oceans, St. John's
52	P. Purych	Ontario Ministry of Natural Resources
53	B. Ruggles	Ontario Hydro
54	H.A. Smith	British Columbia Hydro
55	B. Stedwill	Saskatchewan Power Corporation
56	T. Toner	Nova Scotia Power Corporation
57	D. Wales	Ontario Ministry of Natural Resources
58	T. Watson	British Columbia Hydro
59	D. Windsor	Manitoba Hydro

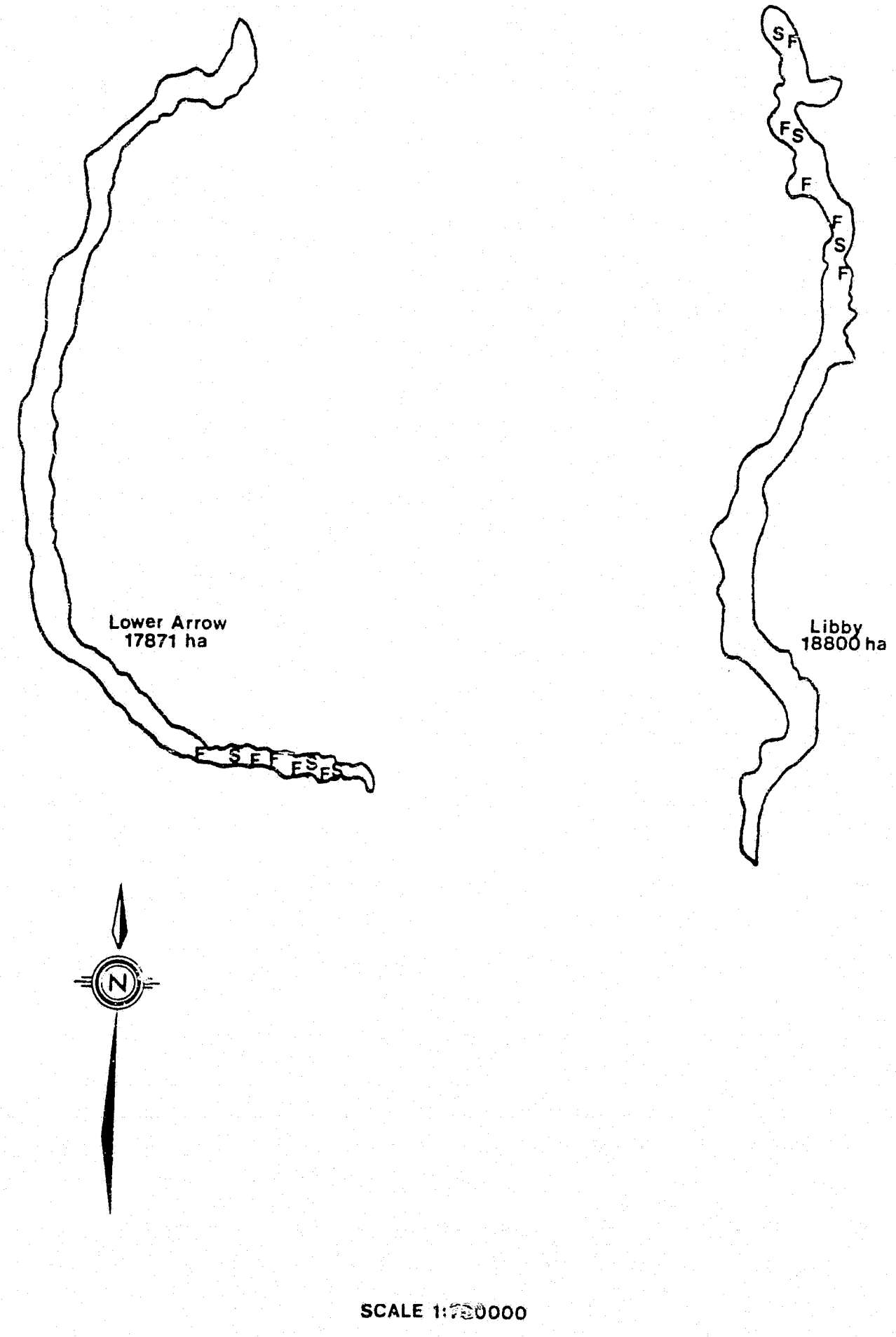
60 Field data collected in 1984

APPENDIX D

SKETCHES OF RESERVOIRS SAMPLED DURING 1984,
SHOWING LOCATIONS OF SEDIMENT (S) AND FISH (F) COLLECTIONS*

* Latitudes and longitudes for these reservoirs are provided in Table 17. A map illustrating their locations is also provided (Figure 8).

BRITISH COLUMBIA



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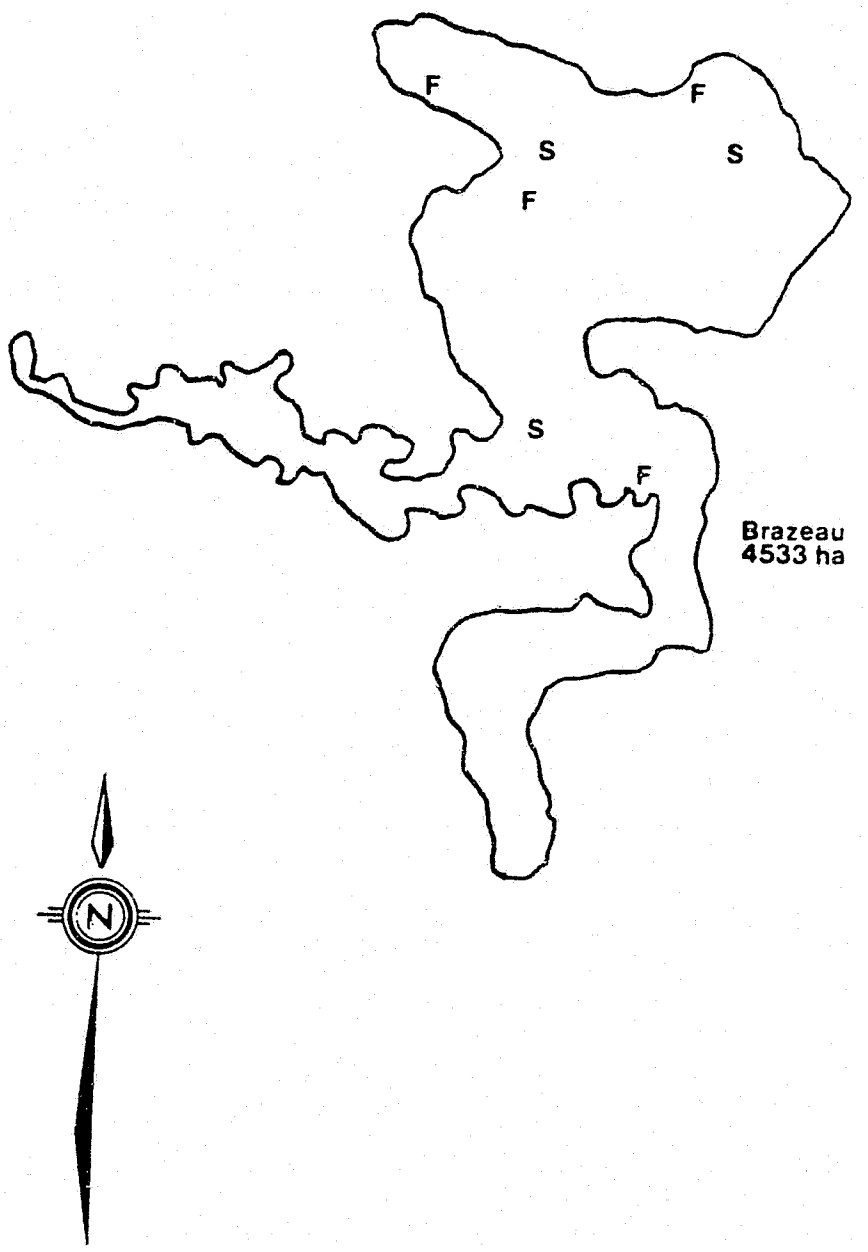
ALBERTA



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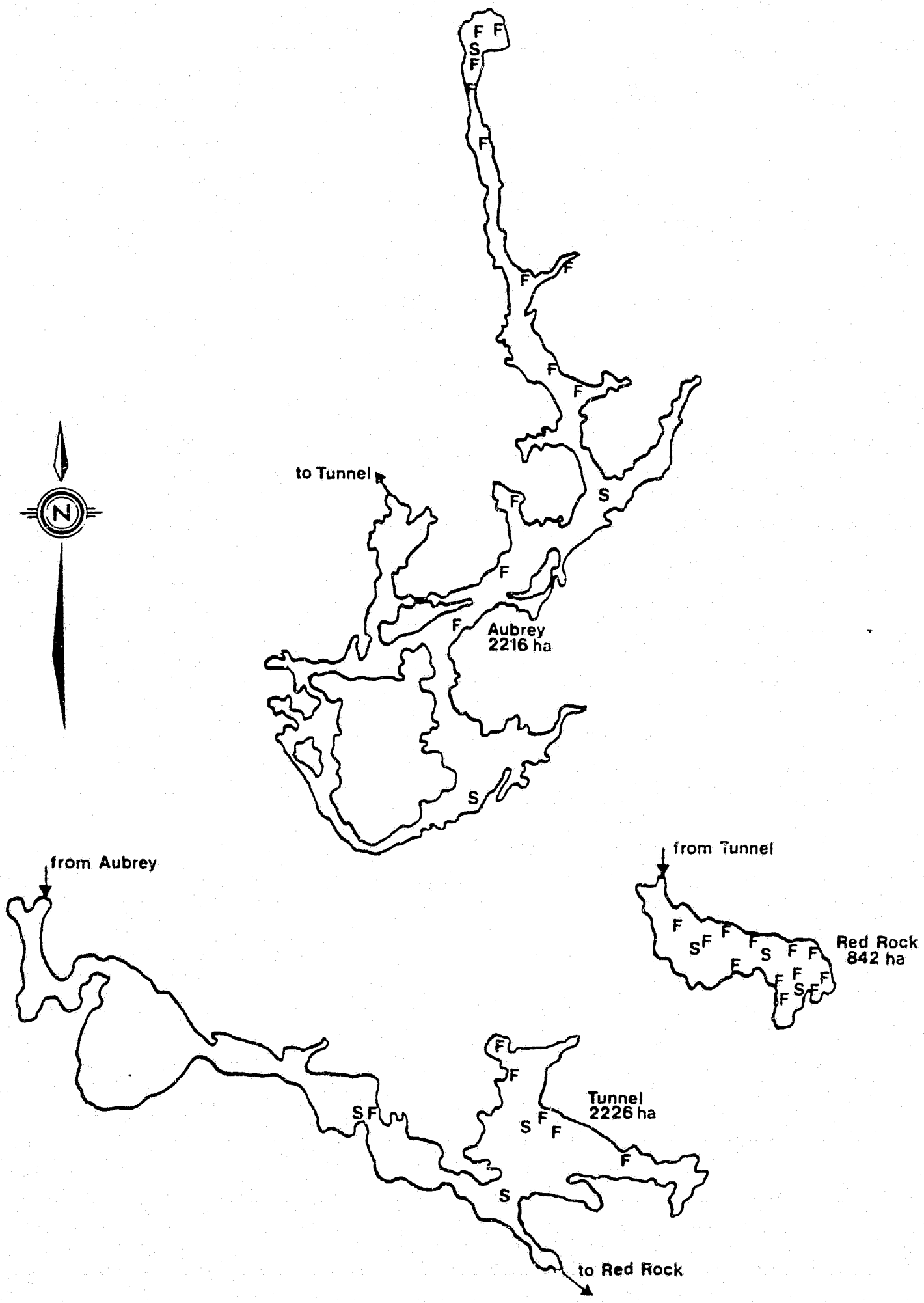
A19

ONTARIO



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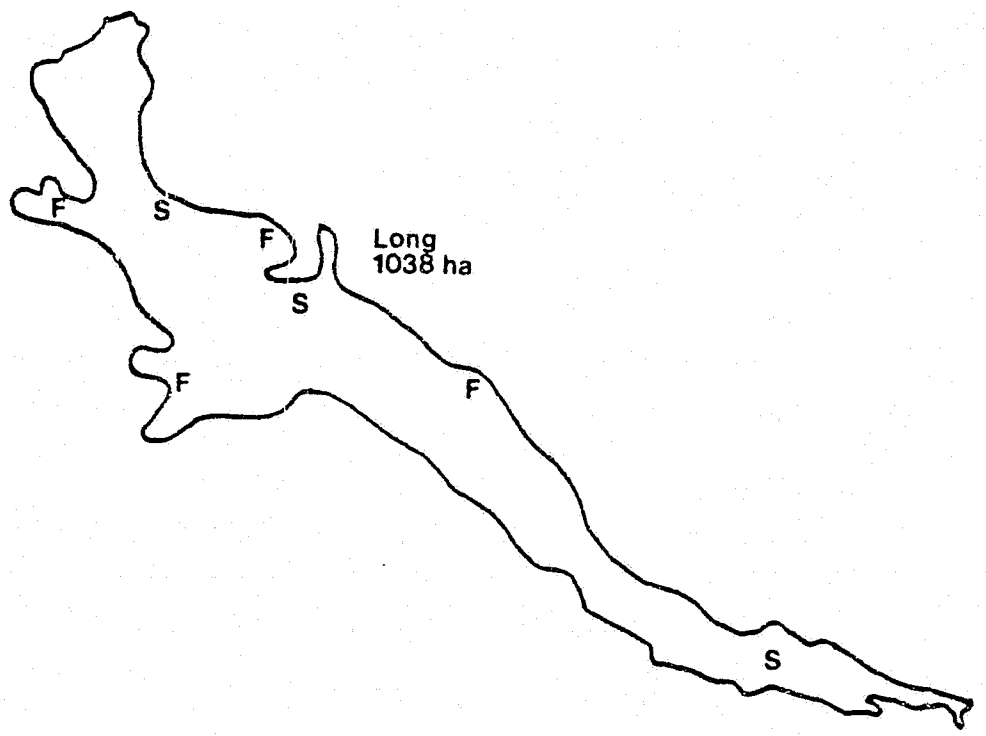
A20



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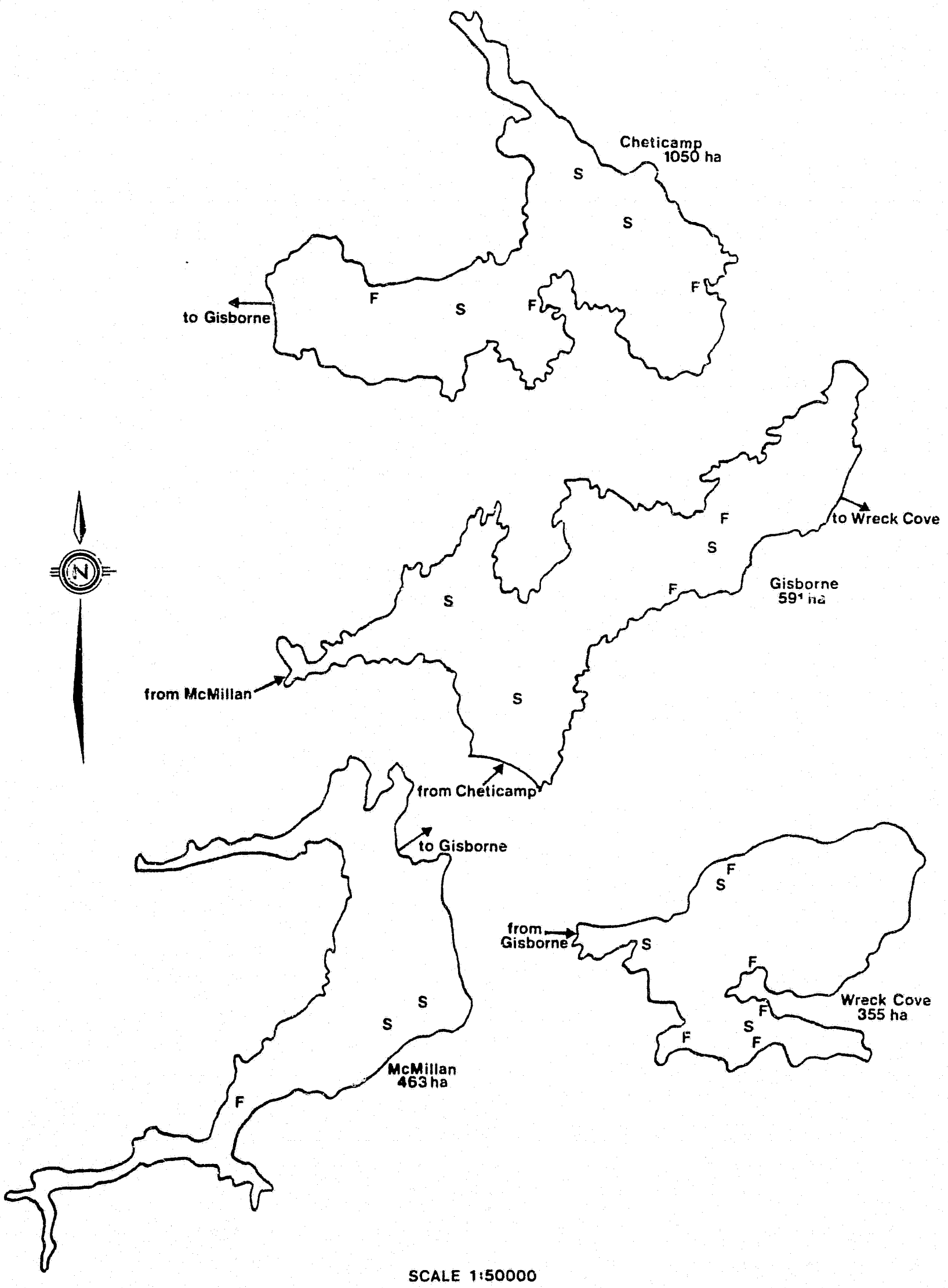
NOVA SCOTIA



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A22