FEDERAL ECOLOGICAL MONITORING PROGRAM

Technical Appendices

VOLUME I

Environment Canada / Environnement Canada

Fisheries and Oceans / Pêches et Océans

FEMP TECHNICAL REPORTS

A large number of technical reports were produced under the auspices of FEMP. Most of these reports have already been published; some as part of the FEMP report series, some as part of the DFO technical report series, and some as DOE reports. In addition, there are still some reports that are in preparation. All of these reports are listed in the the following pages to provide the reader with a complete listing of FEMP sponsored studies.

The reports are listed in alphabetical order. The designation of [DOE] or [DFO] after each listing indicates the availability of each report.

For information on DOE designated reports contact

FEMP Reports
Water Resources Branch
Environment Canada
513 - 269 Main Street
Winnipeg, Manitoba R3C 1B2

For information on DFO designated reports contact

FEMP Reports
Department of Fisheries and Oceans
Freshwater Institute
501 University Crescent
Winnipeg, Manitoba R3T 2N6

The five reports published here have not been published previously.

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Baker, G.D. 1990. The Economic Performance of the Southern Indian Lake Summer Commercial Fishery, 1988. M.N.R.M. practicum, University of Manitoba, Winnipeg, Man. 85 p. [DFO]

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SPECIAL NOTE

Conclusions, recommendations or opinions expressed in reports prepared for the Federal Ecological Monitoring Program are those of the authors and do not necessarily represent those of the sponsoring departments.

Federal Ecological Monitoring Program

Final Mercury Report

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Executive Summary

This report summarizes the results of studies on mercury in the area covered by the Northern Flood Agreement conducted under the Federal Ecological Monitoring Program (FEMP) between 1986 and 1989. The primary objectives of these studies were (i) to better define the duration of the elevated fish mercury levels in flooded lakes along the Churchill River diversion and Nelson River and, (ii) to improve understanding of the factors regulating microbial mercury methylation and demethylation processes. The studies also documented mercury concentrations in fish and total and methyl mercury concentrations in water to provide information for consumers.

Concentrations of total mercury in the waters of Southern Indian Lake, and the Rat, Burntwood, and Nelson rivers have not been elevated as a result of CRD or LWR. Mean total mercury concentrations in the reservoirs range between 1.07 and 2.1 ng L^{-1} (as Hg). Although total mercury concentrations in the reservoirs are not elevated, methyl mercury concentrations in water are higher (by 55 to 245%) at reservoir sites over flooded terrain than at non-flooded reference sites. Mean methyl mercury concentrations in water sampled over reservoir sediments supporting elevated net microbial methylation ranged between 0.034 and 0.054 ng L^{-1} (as Hg) compared with 0.022 ng L^{-1} in a non-flooded reference lake. The highest concentrations occurred in the most extensively flooded reservoirs. The elevated concentrations in reservoirs were restricted to sites where net methylation in the sediments was

elevated. No relationship was found between concentrations of total and methyl mercury in water. The increased methyl mercury concentrations in reservoirs of the NFA area do not represent a direct health hazard for those using these waters as a source of drinking water.

Evidence of downstream transport of methyl mercury and mercury contaminated fish, affecting fish mercury levels in waters below hydroelectric reservoirs, has been found in lakes below Notigi Dam and elsewhere in Canada. This report presents evidence which suggests flooding above Kelsey Generating Station on the Nelson River may be affecting fish mercury levels in Split Lake. Additional investigation is recommended to confirm this hypothesis.

The types of bacteria which conduct methylation and demethylation in natural lake and flooded reservoir sediments were identified using selective metabolic inhibitors. Methanogenic and sulphate reducing bacteria jointly accounted for 66% of seasonal mean methylation and 50% of mean demethylation in Granville lake sediments. The two groups were equal contributors to both processes. Conditions in the flooded zone of Methyl Bay appear to be attractive to these groups, where they accounted for all of methylation and demethylation. The proportions of total methylation and demethylation conducted by methanogens in the flooded sediments were about the same as in Granville Lake. The shares of methylation and demethylation in the flooded sediments attributable to sulphate reducers were about double those in Granville Lake.

Surveys of microbial mercury methylation and demethylation rates in sediments from a number of lakes and reservoirs representing a range of physical and chemical conditions were conducted to determine the factors which were most important in regulating net methyl mercury production. The primary limiting factor of methylation is organic matter availability, which influences methylation rates by regulating bacterial activity and by promoting development of the anoxic conditions which are favoured by the methanogenic and sulphate reducing bacteria. Methylation in sediments of lakes and reservoirs in the NFA area is not limited by inorganic mercury availability. Although methylation rates were positively correlated with total mercury concentrations in sediments, this relationship was the product of collinearity with sediment organic content.

The factor or factors which regulate microbial demethylation rates are less well understood, although organic matter and total microbial activity appear to be most important. Demethylation rates were positively correlated with both sediment organic content and total microbial activity. Sediment pH, total mercury concentration, and redox potential had no significant influence on demethylation rates although these variables often were collinear with the significant regulating factors.

An important distinguishing factor between methylation and demethylation is the lack of an influence of redox potential on demethylation rates. This may provide a partial explanation for the differential stimulation of methylation over demethylation in flooded sediments, although additional study on the role of sediment oxygen conditions in regulating demethylation is required.

The expected duration of the elevated fish mercury levels in reservoirs of the NFA area was examined by monitoring microbial rates of mercury methylation and demethylation at flooded and reference sites in the CRD area, monitoring mercury levels in fish from flooded lakes in northern Manitoba, and measuring methylation and demethylation rates in flooded sediments from reservoirs in the boreal forest zone ranging from 24 to 59 years of age.

Six years of methylation and demethylation rate measurements in flooded sediments from Southern Indian Lake and in Granville Lake, an upstream reference lake, indicate no significant declining trend in the elevated net microbial methylation rate in flooded sediments. The differential in net methylation rate, indicated by the M/D ratio, between flooded and reference sediments was highest in 1989.

Mercury levels in whitefish, walleye, and pike from Rat and Threepoint lakes on the CRD, flooded in 1974-1976, and in Stephens Lake on the Nelson River, flooded 1970-1971, were surveyed between 1983 and 1989 by Manitoba Natural Resources, Fisheries Branch. Mercury levels have peaked in all species in all three lakes, and appear to be starting to decline. These declines are more evident in walleye and pike than in whitefish. The rate of return to pre-development mercury levels is projected to be slow, as indicated by the mean concentrations in pike and walleye from Stephens Lake which were 3 to 4 times above pre-development values 18 years after impoundment.

Mercury levels in whitefish and walleye from SIL had peaked in all regions of the lake by 1981 and had declined substantially by 1988, although not to pre-development levels. In contrast, mercury concentrations in pike had not started to decline as late as 1988, 12 years after impoundment.

The different temporal trends in whitefish and pike in SIL compared with Rat and Stephens lakes were attributed to two factors; (i) the interaction of the habitat preferences of these species with the relatively small extent of flooding on SIL compared to the other reservoirs and, (ii) the changes in limnological conditions in which have occurred in Notigi Reservoir and Stephens Lake. The declines in pike mercury levels in Rat and Stephens lakes appear related to a recent increase in shoreline erosion rates. The consequent increase in nearshore sedimentation may be reducing the area of exposed organic matter and, therefore, the area of reservoir sediments supporting elevated net methylation rates. The resulting decrease in methyl mercury availability produces lower mercury concentrations in pike. Nearshore sedimentation would not affect whitefish in these lakes because of their preference for the pelagic zone.

The greater decrease in whitefish mercury concentrations in SIL than in Rat or Stephens lakes since the initial increase following impoundment appears to be due to the offshore habits of this species and the small extent of flooding in SIL. Immediately after flooding, deposits of terrestrial organic matter in SIL extended well beyond the margin of the pre-existing lake shore as a result of offshore transport processes. While exposed, the organics would have stimulated microbial methylation in the offshore zone, contributing to mercury levels in the whitefish. The organics have since been covered by deposits of inorganic sediments, reducing offshore methylation rates to baseline levels with a consequent reduction in mercury levels in whitefish. In the much more extensively flooded Rat and Stephens lakes, whitefish are exposed to flooded sediments which support elevated net methylation rates despite their preference for the pelagic zone.

The fish mercury data for lakes on the CRD and Nelson River indicate declines are occurring in some species at some locations and these declines are proceeding in a slow and discontinuous manner. With the short data record relative to the observed rate of decline, it is difficult to base predictions of duration on the available fish mercury data. Some other approach is necessary, and measurements of the microbial methylation balance in older reservoirs appear to be the best available method.

Methylation and demethylation rates were measured in flooded sediments from reservoirs ranging from 24 to 59 years of age during the 1986-1989 period of the FEMP study in addition to annual measurements in SIL. Elevated methylation rates were found in flooded sediments from all reservoirs. Elevated demethylation rates were found in reservoirs up to 43 years of age. The elevated methylation and demethylation rates both declined over time following negative exponential decay curves. Reservoir age accounted for 64% of the variation in methylation rates and 78% of the variation in demethylation rates. Methylation and demethylation rates were projected to return to baseline 78 and 53 years, respectively, after impoundment. Net methylation, indicated by the methylation balance (M/D), declined following a second order polynomial. Extrapolation of this curve to M/D in reference lakes indicated the elevated methylation balance in SIL, lakes of the CRD, and Nelson River should return to pre-development levels 78 years after impoundment. The projected rate of decline in M/D is consistent with the observed mercury concentrations in pike and walleye from reservoirs with similar morphology and biophysical setting ranging from 12 to 47 years of age.

Factors which can affect the duration of a reservoir mercury problem include the nature of the flooded organic matter with respect to its ease of decomposition, the presence of a continuing source of fresh organic matter, and the erosion of shoreline clays. The duration of the problem in the CRD is based on the sampling of reservoirs in which boreal forest was flooded. This vegetation is resistant to decay, leading to the lengthy duration of the problem. If more readily decomposed vegetation is flooded, mercury levels in fish return to baseline much more quickly. This was the case in Cedar Lake, Manitoba, where the majority of flooded land was vegetated by marsh macrophytes and sedges and where there was no evidence of elevated fish mercury within 14 years of impoundment.

The presence of a continuing source of fresh organic matter appears to be important in maintaining fish mercury levels in

Sipiwesk Lake. The lake has frequently been subjected to low water levels in summer since construction of the Jenpeg Dam, with exposure of the drawdown zone during the growing season permitting considerable vegetation growth. This fresh organic matter is introduced to the lake during periods of high water and stimulates microbial methylation. It is expected this process will maintain the high fish mercury levels in Sipiwesk Lake for an indefinite period.

The occurrence of declines in fish mercury levels in Rat and Stephens lakes coincident with increases in shoreline erosion suggest erosion of inorganic clays may have an influence on the rate of recovery of reservoir mercury problems. Deposition of the eroded clays appears to be reducing the area of reservoir sediments supporting elevated net methylation rates. The long-term effect of this process on the rate of recovery of the mercury problem remains to be established.

The potential effectiveness of several measures for the mitigation of aquatic mercury problems or for the rehabilitation of fisheries closed due to elevated mercury levels was reviewed. Methods considered included selenium additions, increased water column turbidity, reservoir liming, inhibition of microbial methylation or stimulation of demethylation, increasing fish growth rates through intensive fishing or nutrient additions, and the selective harvest of smaller individuals. Few of these methods appear to have practical application to the reservoir mercury problem. Selenium additions can be effective in reducing mercury bioaccumulation by fish but the concentrations which must be maintained are very close to the Canadian limit for drinking water. This method is unlikely to be accepted for general application because of the high concentrations which must be maintained and, unless the effective level can be substantially reduced or it can be shown that the required level of addition is completely non-toxic, the method will be unacceptable for application to any problem. Limnocorral experiments demonstrated that increased water column turbidity is not an effective means of reducing mercury bioaccumulation by fish. Reservoir liming will be of no use in the NFA area because the elevated net methylation rates are not due to a reduction in pH. Increasing fish growth rates through nutrient additions also has no application in the NFA area because primary productivity in most of the turbid reservoirs is limited by light rather than by nutrients. The development of measures to directly manipulate the methylating and demethylating bacteria does not appear to be of any immediate use, requiring considerable additional research. In contrast, the use of intensive fishing to reduce fish mercury levels and the implementation of a size-selective commercial fishery to make use of smaller individuals of the higher value species with lower mercury concentrations both may have some limited application in the NFA area. Additional investigation is required to identify the benefits and limitations of these approaches.

The cause of the reservoir mercury problem, stimulation of microbial methylation by flooded organic matter, suggests a number of possible measures for the minimization or prevention of elevated fish mercury levels in new reservoirs. These methods include mechanical clearing of organic matter from land to be flooded, combustion of organic matter, selective removal (mechanical or combustion) of the types of organic matter which most contribute to elevated net methylation, and isolation of flooded organics from the water column by covering with inorganic soils. The only approach which can be recommended based on existing knowledge is the complete removal of all organic matter, including organic soils, prior to flooding. Further research is necessary to assess the cost, environmental impact, and effectiveness of the other approaches.

The extensive database for reservoirs in the NFA area has been employed in several attempts to develop numerical models for prediction of the severity of fish mercury problems in new reservoirs based on their physical and chemical properties. Early efforts were unsuccessful due to their failure to consider the influence of upstream reservoirs and temporal trends in fish mercury levels. Current models which incorporate these variables account for 69-84% of the variation in whitefish, walleye, and pike mercury burden in flooded lakes on the CRD. These models should be useful for prediction of the increase in fish mercury levels expected in new reservoirs planned for the CRD. Additional investigation is required to modify the models for use in other regions.

Examination of the effect of the proposed Cross Lake weir on fish mercury levels in the lake indicated that mercury levels in fish are not expected to rise following construction. The weir will raise the minimum water level without increasing high water levels above the historical maximum and no back-flooding will occur. The flooding of the re-vegetated drawdown zone of the lake also is not expected to increase mercury levels in fish. Lake levels have fluctuated considerably since construction of the Jenpeg Dam, yet mercury levels in fish have not become elevated above natural levels for the region. This appears to be due to the bedrock controlled shoreline of Cross lake which limits the re-growth of vegetation during periods of low water.

Literature on the effects of elevated mercury levels on fish physiology and behaviour was reviewed to determine if the elevated levels along the CRD represent a risk to the health of the fish. The highest muscle mercury levels recorded on the CRD since impoundment approach the upper limit of the no-effects range determined in laboratory studies. This indicates that significant effects on fish physiology and behaviour are unlikely to have occurred. Mercury levels in mink and otter from the CRD area also are generally within the no-effects range.

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1.0 Introduction

The occurrence of elevated mercury concentrations in fish from Southern Indian Lake (SIL) and other lakes flooded by the Churchill River diversion (CRD) is a continuing source of concern for the health of the Cree and other northern populations in the area covered by the Northern Flood Agreement (NFA) which depend upon fish as a staple food resource. This mercury problem was not a predicted impact of the Churchill River diversion, yet it is among the most severe of the impacts which have been documented (Hecky et al. 1984). Consequently, mercury has been the subject of considerable study since the problem was first identified in 1979 (Bodaly and Hecky 1979); initially under the Canada-Manitoba Agreement on the Study and Monitoring of Mercury in the Churchill River Diversion (CMMA) and subsequently as part of the Federal Ecological Monitoring Program (FEMP) undertaken by Environment Canada and the Department of Fisheries and Oceans (DFO), and the Ecological Monitoring Program (EMP) conducted by the Province of Manitoba. This report summarizes the findings of studies on mercury conducted by these agencies under the FEMP and the EMP, with emphasis on investigations sponsored by Environment Canada. Detailed reports of the Environment Canada studies can be found in Ramsey (1987a, 1988a, 1989, and 1990). Green (1990) and Ramsey (1991) present and analyze fish mercury data gathered under the EMP. Strange et al. (1991) present and analyze fish mercury data from SIL and Issett Lake gathered by the Department of Fisheries and Oceans.

The studies undertaken by Environment Canada concentrated on examination of the **microbial** processes which regulate **monomethyl mercury** levels in the aquatic environment. Monomethyl mercury, referred to simply as methyl mercury in this report, is important because it is the only form of mercury which accumulates in fish. **Bacteria** produce methyl mercury from inorganic mercury in the process of **methylation.** Methyl mercury is decomposed to elemental mercury and methane in the process of **demethylation** which also is conducted by bacteria. The balance between the rates of methylation and demethylation regulates the availability of methyl mercury and, in turn, determines the concentration of mercury in fish tissues. These processes are discussed in more detail in Section 1.2 of this report.

Surveys of methylation and demethylation rates in the CRD conducted under the CMMA found elevated net methylation rates in sediments taken from areas of flooded terrain but not in sediments from the pre-existing lake basins (Ramlal et al. 1987; Ramsey and Ramlal 1987 a and b). Rates of both methylation and demethylation were elevated in flooded sediments, but methylation was differentially stimulated resulting in the elevated net methylation rate. **Limnocorral** experiments indicated the most important factors regulating net mercury methylation were the amount and type of flooded organic matter (Hecky et al. 1987a).

Early studies of fish mercury levels in reservoirs located in the central and southern United States suggested the elevated levels were transient and would return to baseline levels 3 to 5 years after impoundment (Cox et al. 1979; Abernathy and Cumbie 1977). The CMMA studies found this estimate was not applicable to northern reservoirs. Based on the observation of elevated net methylation in the flooded zone of a 40 year old reservoir and the lack of a significant downward trend in mercury levels in predatory fish species from the CRD, it was speculated that several decades would be required for recovery (Hecky et al. 1987b). A more precise estimate of the duration was not possible, however, due to the lack of information from older reservoirs in northern Canada.

The CMMA studies also found there was no proven means of mitigating reservoir mercury problems. It was suggested that an area of investigation which might be followed was the manipulation of the methylating or demethylating bacteria in reservoirs to reduce net methyl mercury production. This approach required greater knowledge of the causative micro-organisms and their limiting factors than was available at the start of the FEMP. Thus, the primary goals of the FEMP mercury program were:

i). to better define the duration of the mercury problems in flooded lakes along the Churchill River diversion and Nelson River, and;

ii). to provide a better understanding of the factors regulating methylation and demethylation to determine if it is possible to regulate methyl mercury production in reservoirs.

Two approaches were used in assessing duration. Methylation and demethylation rates were measured annually, from 1986 to 1989, at a site on SIL and in Granville Lake (Figure 1), a non-flooded reference lake, to look for evidence of a decline in the methylation balance in flooded sediments over the short term. The SIL site was located in Methyl Bay, a small protected bay near the community of South Indian Lake (Figure 1), and included stations in flooded terrain and in the pre-existing lake basin. The same measurements were made in sediments from the flooded zones of a number of older reservoirs in Manitoba, Saskatchewan, (Figure 1) and northwestern Ontario, ranging from 24 to 59 years of age, to examine long-term trends in methylation and demethylation activity. These sites represented a broad range of physical and chemical conditions, enabling examination of the significance of various environmental factors in the regulation of microbial methylation and demethylation. The groups of bacteria responsible for methylation and demethylation in SIL and Granville Lake also were identified using specific metabolic inhibitors.

Studies of mercury on the Churchill River diversion and Nelson River have largely concentrated on measurements of mercury levels in fish, and on examination of the processes involved in transformations of mercury in the aquatic environment. These studies were designed to serve the information requirements of the various government agencies with responsibilities in the area and to satisfy the perceived information needs of the affected communities. However, it is possible that some issues of importance to

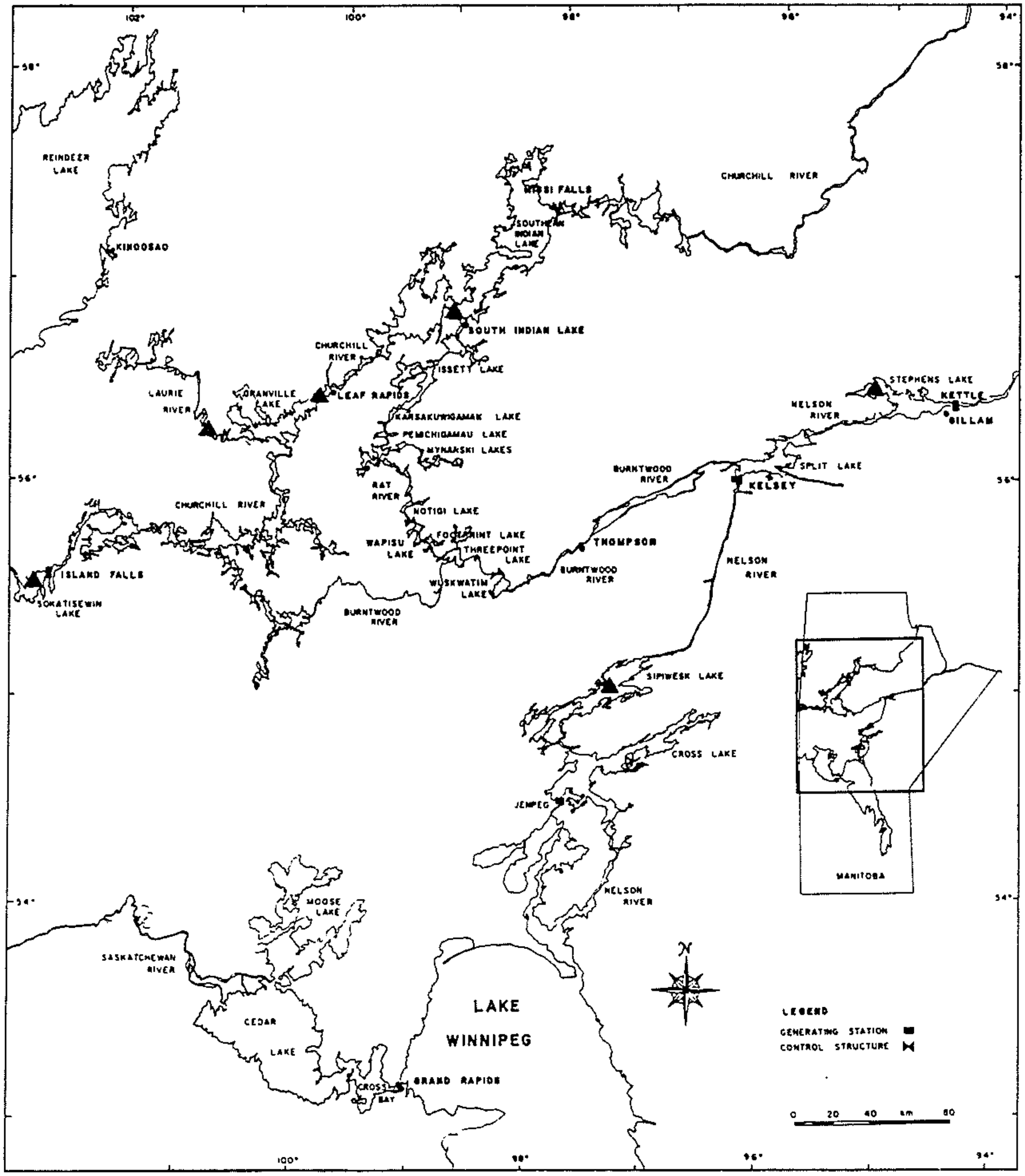

Figure 1. Location of methylation balance study sites (▲) in northern Manitoba, 1985-1989.

specific communities remain to be identified and examined. In this regard, a survey of community opinion with respect to outstanding mercury-related issues was undertaken. This survey was by no means exhaustive, as it was not possible to directly solicit public views through meetings, workshops, or community surveys. Instead, people in the communities with a good understanding of the community information requirements, and a position in which they would become aware of mercury-related issues through their daily activities, were interviewed. These people generally were the communities' key communicators. Information from the survey of communities was augmented with information from interviews with representatives of government agencies, consultants working in the area, and Manitoba Hydro. The survey was restricted to non-medical issues.

1.1 Outstanding Issues

One issue of particular importance identified by several of the people contacted is the problem of translating the word mercury into the Cree language. In the CMMA information program, mercury was interpreted as "something like a poison, in that too much is harmful" (L. McKerness, Environment Canada, Winnipeg, MB. pers. comm.). This definition is problematic in that the word poison seems to have become the focus of attention. There is a general anxiety regarding mercury levels in fish throughout the NFA area, including the Cross Lake community where mercury levels in fish are low. It has since been suggested that the term "fish sickness" be used. This also is an imprecise translation because it does not convey the relationship between the amount of mercury in a fish and the relative hazard to health and it implies that the fish become sick as a result of high mercury levels, which does not seem to be the case (Section 5.0). Any attempt to communicate the conclusions of this and other mercury studies in Cree will be limited by the language barrier yet there is the risk that an incomplete translation may cause unnecessary anxiety. In the absence of a satisfactory translation, it has been suggested that the english word, mercury, be used. This approach is not entirely satisfactory but may help to prevent misunderstanding related to imprecise translation.

There also seems to be a need for the presentation of information on the mercury issue in a manner that is understandable to people without a technical or scientific background. The communication of information on mercury and other issues to the NFA communities has been a high priority of the FEMP. However, my discussions with the communities indicate the program has not been entirely successful. More interactive approaches appear to be necessary to promote fuller understanding of the reservoir mercury problem. These would require active community participation, perhaps in the form of workshops dealing with specific issues.

The question of the effect of the elevated mercury levels in fish and other animals on the health and behaviour of the animals also was raised. Only one of the studies conducted under the CMMA mentioned toxicity to wildlife (Kucera 1987), and none of the studies conducted thus far has examined the potential significance of mercury-related health effects on fish and wildlife in any detail. Section 5.0 in this report summarizes current knowledge on the effect of elevated mercury levels on the behaviour and health of fish and aquatic mammals.

Among the major outstanding questions with regard to mercury in the Split Lake area is whether the upstream hydroelectric developments on the Nelson River and Churchill River diversion have affected in the past, or are continuing to affect, mercury levels in fish from the lake (A. Beardy, Split Lake, MB, and W. Wysocki, Symbion Consultants, Winnipeg, MB, pers. comm.). This question is considered in Section 2.4.1.

Another concern of Split Lake residents is the lack of information on mercury levels in biota other than fish. The "country" food base of the community includes many products of fish-eating birds (e.g., gull eggs and mergansers) and mammals (e.g., bear) for which no information has been published.

The proposed weir near the outlet of Cross Lake for the maintenance of water levels also has raised the question of possible effects on mercury levels in fish (L. McKerness, Environment Canada, Winnipeg, MB, pers. comm.). This concern would appear to be a product of experiences in other northern communities where mercury levels in fish have increased after a rise in water levels. The potential effect of this weir on fish mercury levels in Cross Lake is evaluated in Section 8.1 of this report.

1.2 Mercury in the Aquatic Environment

Mercury is a naturally occurring metal and can be found, in varying levels, in all soils, water, and biota. The ultimate source of all mercury in the environment is the inorganic mercury in bedrock (Andersson 1979). Although originally derived from this source, the mercury found in fish occurs in an organic form called monomethyl mercury. Long considered to be the dominant form of mercury in fish muscle tissue (Huckabee et al. 1979), recent research has shown that methyl mercury is the only form which accumulates in fish muscle (Bloom 1989). Any other forms which have previously been reported as occurring in fish muscle are the product of sample contamination in the field or laboratory.

In aquatic systems, methylation of inorganic mercury by bacteria is the only important means of methyl mercury production. Abiotic, or chemical, methylation also has been shown to occur, but is thought to account for less than 10% of total methyl mercury production (Jensen and Jernelov 1969; Furutani and Rudd 1980; Berman and Bartha 1986). Once produced, methyl mercury is very stable in the natural environment. The only known abiotic mechanism of methyl mercury decomposition is **photolysis**. However, the low sunlight absorption rate constant for the **methyl mercuric ion (CH_3Hg^+)** is thought to preclude photodecomposition as a significant degradation pathway (Baughman et al. 1973). The only other known natural means of decomposition is demethylation by bacteria (Bisogni 1979).

Methyl mercury is of particular concern for human health because of its ability to cross the blood-brain barrier and concentrate in the central nervous system (Carty and Malone 1979), causing a variety of nervous disorders. Methyl mercury also has a strong tendency to accumulate in higher animals, such as fish, furbearers, and humans, because of its strong affinity for **sulfhydryl groups** in **proteins** (Carty and Malone 1979). The report by Health and Welfare Canada (1987) can provide additional information on the effects of methyl mercury on human health and the levels which place a person at risk of health impairment.

1.2.1 Mechanisms of Microbial Methylation and Demethylation

The mechanism of biological methylation remains to be fully described but, at present, is thought to involve the **nonenzymatic** transfer of methyl groups (CH_3^-) from **methylcobalamin (Vitamin B_{12})** to **Hg^{2+} (mercuric ions)** in bacterial cells (DeSimone et al. 1973; Robinson and Tuovinen 1984). The three major **co-enzymes** involved in methyl transfer reactions in cells are N5-methyltetrahydrofolate derivatives, S-adenosylmethionine, and methylcobalamin. Methylcobalamin is considered to be the co-enzyme responsible for methylation of Hg^{2+} salts because it is the only one capable of transferring **carbanion** methyl groups (Bertilsson and Neujahr 1971; DeSimone et al. 1973). The reaction proceeds by the **electrophilic** attack of the mercuric ion (Hg^{2+}) on the methyl (CH_3) group of Vitamin B_{12} to produce the methyl mercuric ion (CH_3Hg^+) as follows:

$$Hg^{2+} \xrightarrow{CH_3B_{12}} CH_3Hg^+$$

The occurrence of Vitamin B_{12} in most bacteria and the lack of need for a specific **enzyme** to facilitate transfer of the methyl group means that many different types of bacteria are capable of mercury methylation. It is not a specialized process, rather it can be viewed as an accident of normal cell metabolism. Although non-enzymatic methylation appears to be the most common mechanism, enzymatic transfer of methyl groups (CH_3^-) to mercuric ions (Hg^{2+}) also has been suggested to occur as some bacteria lacking methylcobalamin have been shown to methylate mercury (Landner 1971).

Demethylation is the better understood of the mechanisms regulating methyl mercury availability. The process has two enzyme-mediated steps: (1) the cleavage of the carbon-mercury bond of the methyl mercuric ion (CH_3Hg^+) by **organomercurial lyase** to produce methane (CH_4) and a mercuric ion (Hg^{2+}); and, (2) the reduction of Hg^{2+} to elemental mercury (Hg°) by **mercuric reductase** (Robinson and Tuovinen 1984; Summers and Silver 1978) as follows:

$$CH_3Hg^+ \xrightarrow[\text{organomercurial lyase}]{NADPH \;\searrow\; \nearrow\; NADP^+} CH_4 + Hg^{2+} \xrightarrow[\text{mercuric reductase}]{NADPH \;\searrow\; \nearrow\; NADP^+} Hg^\circ$$

The requirement for two specialized enzymes makes the ability to demethylate a trait found only in those bacteria which carry the genetic information required to produce these enzymes. The genetic information for production of organomercurial lyase is encoded on transposons (sexually transmitted plasmids; Pan-Hou et al. 1980) rather than on the bacterial chromosome, so that the number of plasmids does not necessarily increase with the size of the microbial community. This is one possible explanation for the differential stimulation of methylation over demethylation in reservoir sediments (Ramsey 1987a).

1.2.2 Methods of Measuring Methylation and Demethylation

The specific rate of mercury methylation was determined in the FEMP studies using the **radioisotope** method developed by Furutani and Rudd (1980) which measures the rate of methylation of $^{203}Hg^{2+}$. The specific rate of demethylation was determined using the radioisotope method of Ramlal et al. (1986) which measures the rate of decomposition of $^{14}CH_3HgI$. Unlike other methods of measuring rates of mercury transformations, both methods used here minimize sample incubation time and the quantity of mercury which must be added so that the microbial community is not altered from that in situ, as indicated by linear methylation and demethylation kinetics over the incubation period (Furutani and Rudd 1980; Ramlal et al. 1986).

The rates derived with each method are termed specific because both processes are dependent upon mercury concentration and, as some mercury is added to the samples, the rates measured are specific to the quantity added. Addition of the same quantity of mercury to all samples should make differences in specific rates between locations proportional to differences in the **in situ** rates (Furutani and Rudd 1980). If concentrations of the bioavailable mercury species (Hg^{2+} for methylation, and CH_3Hg^+ for demethylation) were known, the specific rates could be converted to absolute rates. However, analytical methods for these mercury species in sediments have yet to be developed, restricting expression of the rates to the specific form. The only practical limitation of leaving the measured rates in the specific form is that the difference between the specific methylation and demethylation rates does not yield an absolute net methylation rate. Instead, the ratio of these rates (**M/D**), or what we call the **methylation balance**, is used as an indicator of the relative difference in the net rate of methyl mercury production between locations.

1.2.3 Relationship Between Methylation Balance and Fish Mercury

Traditional methods of measuring mercury methylation rates employ additions of inorganic mercury, incubation periods on the order of several days to weeks, and measurements of methyl mercury yield by gas chromatography (e.g., Jackson 1987). These methods are considered to measure net methylation rates because the amount of methyl mercury measured at the end of the incubation period represents what is left over after the processes of methylation and demethylation have acted. It might be reasonable to assume the same is true for the radiochemical method used to measure specific methylation rates in the FEMP studies. This does not seem to be the case, however, as indicated by an experiment conducted in streams on the Queen Charlotte Islands, British Columbia (Ramsey 1988b).

The experiment involved the use of caged juvenile coho salmon (*Oncorhynchus kisutch* Walbaum) to monitor methyl mercury availability at three locations; Gold Creek, Barbie Wetland, and Middle Barbie Creek. Measurements of microbial methylation and demethylation were made at the same sites using the radioisotope methods described above.

Mercury uptake by the fish was the same at two of the sites, Gold Creek and Barbie Wetland (Figure 2). Uptake at these sites was significantly higher than at Middle Barbie Creek. The specific methylation rates in Gold Creek and Barbie Wetland sediments also were significantly higher than at the Middle Barbie Creek site, but the specific methylation rate in Barbie Wetland was significantly higher than in Gold Creek (Figure 2). M/D in Gold Creek and Barbie Wetland was significantly higher in Middle Barbie Creek and there was no significant difference in M/D between Gold Creek and Barbie Wetland. Methyl mercury uptake by the fish was proportionate to the methylation balance (M/D) but not to the specific methylation rate (Figure 2). This indicates M/D is a measure of the net methylation rate while the specific methylation rate is not, contrary to traditional analytical methods. All studies of microbial methylation and demethylation conducted under the CMMA and FEMP which have used the radiochemical analytical methods have employed

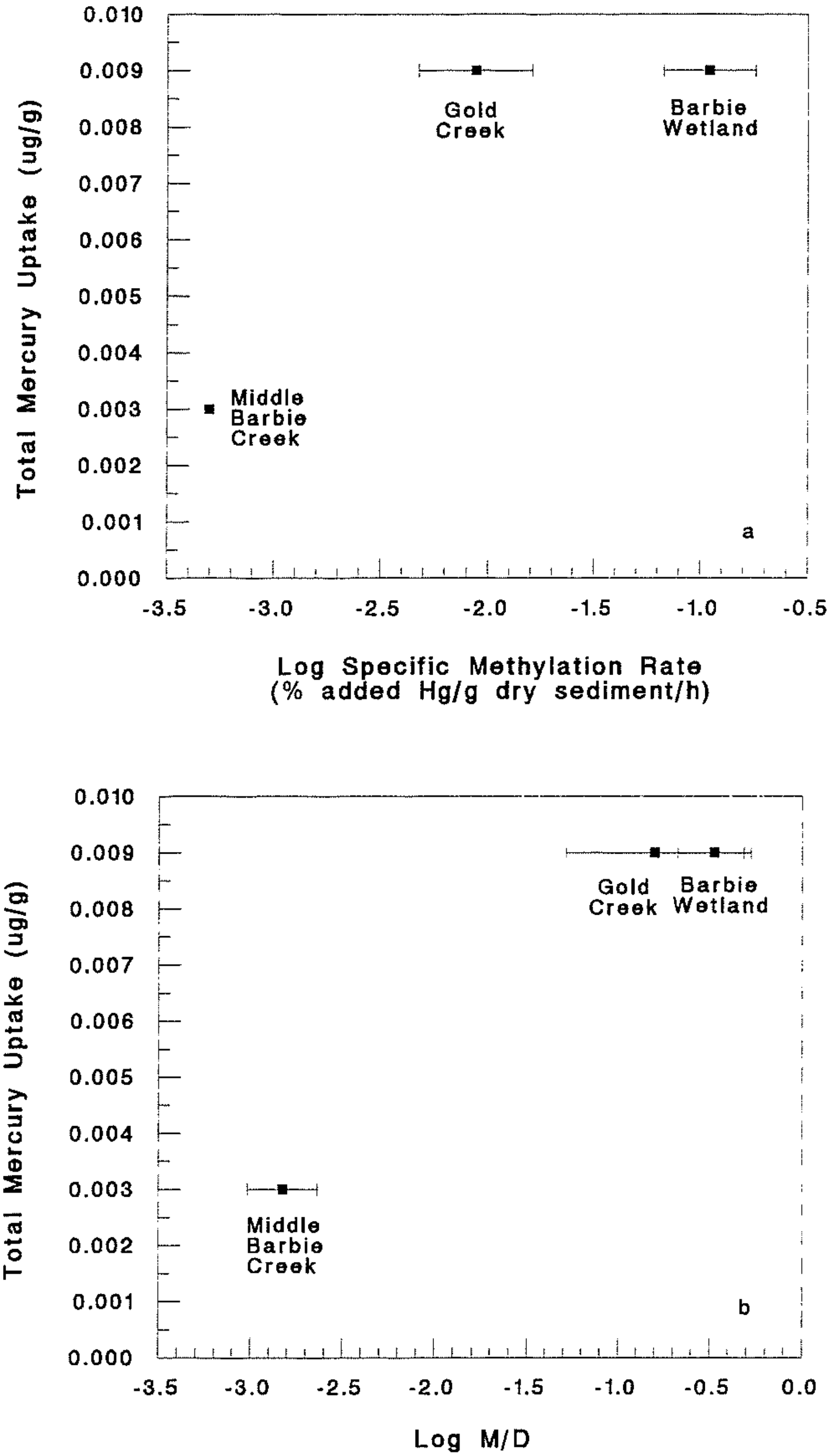

Figure 2. Relationship between mercury uptake by caged juvenile coho salmon and (a) the specific methylation rate, and (b) the methylation balance (M/D) in stream sediments on the Queen Charlotte Islands, British Columbia. Fish mercury data from G. Derksen, Environment Canada, Vancouver, BC. Methylation and M/D data from Ramsey (1988b).

M/D as a measure of the net methylation rate.

For a methylation rate measured using the radiochemical method to be a net value, **isotopic equilibrium** between the radio-labelled methyl mercury produced during the incubation and the non-labelled methyl mercury, both present in the sediments at the start of the assay and produced during the assay, must be attained early in the incubation period. With a 12 hour incubation period, and very slow methylation rates (between 0.0005% and 0.03% added Hg (g dry sediment)$^{-1}$ h^{-1}), it is unlikely that equilibrium is approached during the assay. The radiochemical method probably does not yield a true gross methylation rate, as some of the radio-labelled methyl mercury produced in the assay undoubtedly is demethylated before the end of the incubation. However, the failure of the methylation rate alone to account for mercury uptake by the caged fish indicates the rate is much closer to the gross than to the net value.

1.2.4 Identity of Methylating and Demethylating Bacteria

The methylating and demethylating bacteria were identified, at the level of major metabolic group, using selective metabolic inhibitors. These inhibitors prevent a specific type of bacteria (e.g., **methanogenic**) from conducting the chemical reactions necessary for growth and reproduction without substantially affecting the activity of other types of bacteria. Two groups of bacteria, **methanogens** and **sulphate reducers**, were singled out for study as other investigations had shown they were the most likely to be important in mercury transformations (Wood et al. 1968; Compeau and Bartha 1985). Bromoethane sulfonic acid (BESA), a specific inhibitor of **methanogenesis** (Gunsalus et al. 1978), was used to examine the role of methanogenic bacteria in methylation and demethylation. Sodium molybdate, a specific inhibitor of sulphate reduction (Postgate 1949; Saleh et al. 1964), was employed to identify the importance of sulphate reducing bacteria in these processes. The importance of each group was determined by measuring the reduction in the specific methylation and demethylation rates caused by the addition of each inhibitor. This aspect of the study was restricted to Methyl Bay and Granville Lake. Details of the method are presented in Ramsey (1989a).

The methanogens and sulphate reducers were found to jointly account for the majority of seasonal mean methylation (66%) and demethylation (50%) in Granville Lake sediments (Table 1). The two groups were equal contributors to both processes. Conditions in the flooded zone of Methyl Bay appear to be particularly attractive to these groups, where they jointly accounted for all of methylation and demethylation. The share of total methylation and demethylation attributable to the methanogens in flooded sediments was the same as in Granville Lake. The shares of total methylation and demethylation attributable to the sulphate reducers in Methyl Bay sediments were approximately double those in Granville Lake. The rates of methylation and demethylation by methanogens in Methyl Bay were a factor of 5 higher than in Granville Lake. Methylation and demethylation rates by sulphate reducers were a factor of 10 higher

Table 1. Comparison of seasonal mean specific rates of microbial methylation and demethylation attributable to methanogenic and sulphate reducing bacteria; July 4 through August 24, 1988. All values are time-weighted means from Ramsey (1989a).

	% added Hg g^{-1} h^{-1}					
	Methylation			Demethylation		
Site	Total	Methanogen	Sulfate Reducers	Total	Methanogen	Sulfate Reducers
Methyl Bay						
1	0.0034	0.0015	0.0029	0.2620	0.0939	0.1404
2	0.0054	0.0026	0.0033	0.3313	0.1862	0.2034
3	0.0042	0.0003	0.0023	0.1004	0.0127	0.0409
Mean	0.0043	0.0015	0.0028	0.2279	0.0976	0.1282
Granville Lake						
5	0.0010	0.0000	0.0006	0.0861	0.0137	0.0175
6	0.0009	0.0008	0.0001	0.0438	0.0140	0.0034
7	0.0007	0.0001	0.0002	0.0523	0.0221	0.0190
Mean	0.0009	0.0003	0.0003	0.0581	0.0166	0.0133

in the flooded sediments.

Both groups of bacteria are **obligate anaerobes** (Mitchell 1974). The increased absolute rates of methylation and demethylation by these groups, and the increased importance of sulfate reducers in these processes are probably a result of the environmental conditions which develop in the flooded soils and vegetation after flooding. The high rates of **aerobic** decomposition in these organic rich sediments could be expected to create anoxic micro-environments and, in extreme cases, complete anoxia in the sediment pore-waters and near-bottom waters. Methanogenesis requires the complete absence of oxygen while sulphate reduction is less strict in this requirement (Mitchell 1974).

The importance of these groups in methylation explains the finding of higher rates in anoxic sediments along the CRD (Ramsey and Ramlal 1987a; Jackson 1987). Recent research suggests methylation rates are highest in the zone of transition between oxic and anoxic conditions (Winfrey and Rislove 1990). This is consistent with the identity of the methylating organisms, and with the chemistry of inorganic mercury.

An inverse relationship between oxygen availability and methylation rate has been frequently reported (Beijer and Jernelov 1979), but it was difficult to understand how this could occur based on the chemistry of inorganic mercury. The sulphides (S^{2-}) produced during sulphate reduction would be expected to bind with mercuric ions (Hg^{2+}) forming cinnabar (mercuric sulphide, HgS). Under anoxic conditions, cinnabar is not available for methylation

(Beijer and Jernelov 1979). This paradox is avoided in the transitional zone. Sulfate reduction can occur, but inorganic mercury is not rendered completely unavailable.

2.0 Mercury in Water

2.1 Total Mercury

The discovery of elevated mercury concentrations in fish from Southern Indian Lake and the Churchill River diversion suggested the that elevated mercury concentrations might be found in the water as well. The fish were accumulating their increased mercury burdens from the water, both directly over the gills and indirectly in their food. Thus, it seemed logical to look for mercury in the water both as an indication of the cause of the problem in fish as well as to determine if the quality of water for human consumption and the maintenance of aquatic life had been affected.

Several approaches and a variety of analytical methods have been applied to the problem of measuring mercury concentrations in water since 1981. Estimates of concentrations have varied widely among the studies, but this variation was largely a result of inadequate sampling methods which introduced contamination. Despite this variability, all studies have been consistent in concluding that concentrations in the flooded lakes are not elevated in comparison to sites which have not been affected by Churchill River diversion.

Bodaly et al. (1984) reported total mercury concentrations from 17 locations on Southern Indian, Issett, Pemichigamau, East, Central, and West Mynarski, Notigi, and Footprint lakes in samples collected in September, 1978, and July, 1981. All values were below the analytical detection limit of 5 ng L^{-1}. These measurements indicated total mercury levels in water were very low relative to guidelines for drinking water, where the upper limit for consumption is 1,000 ng L^{-1} (Health and Welfare Canada 1989), and for protection of aquatic life. The Canadian water quality guideline for protection of aquatic life is 50 ng L^{-1} as total mercury (CCREM 1987). The Manitoba guideline for protection of aquatic life is 6 ng L^{-1} as total mercury (Williamson 1988).

Although the levels were low, it could not be determined if mercury concentrations in water had increased as a result of impoundment and diversion because all values were below the detection limit. It was possible that mercury concentrations had increased by as much as a factor of 5 over the pre-development level, yet still remained below the detection limit of available methods. **Isotope** studies in limnocorrals indicated actual total mercury concentrations might be as low as 1 ng L^{-1} (Hecky et al. 1987a). Information on the magnitude of an increase in mercury levels in water, if any had occurred, was still needed and investigators turned to measurements of total mercury in **net-plankton** as an indirect means of addressing the question. Net-plankton were selected as a surrogate for measurements in water based on research which indicated **plankton** accumulated total mercury in proportion to the concentration in water (reviewed by Ramsey and Ramlal 1987a). The plankton therefore represented a natural pre-concentration step in the analytical procedure.

Most measurements of mercury concentrations in plankton have indicated that total mercury concentrations in water of SIL, the CRD, and Sipiwesk and Stephens lakes on the Nelson River are not elevated in comparison to levels in the Churchill River upstream of SIL (Ramsey and Ramlal 1987 a and b; Ramsey 1987a, 1988a). Concentrations in plankton from the reservoir sites were consistently lower than in reference lakes like Granville and East Mynarski (Figure 3). There are no pre-development net-plankton mercury data for the flooded lakes, so measurements in non-flooded reference lakes provided the next best indication of baseline levels.

The conclusion of the net-plankton studies was confirmed by direct measurement in 1987, when changes in sample collection procedures enabled the reduction of detection limits. Total mercury concentrations in water were positively correlated with concentrations in net-plankton (Figure 4), and the mercury concentrations in water from SIL and Notigi Reservoir were lower than in Granville Lake, upstream of the flooding (Table 2).

The range of total mercury concentrations in waters from the NFA area is summarized in Table 2. There is considerable variance in reported levels with a general trend of decreasing concentrations over time. This decline was a result of improvements of sampling and analytical methods rather than representing a temporal trend. The potential for contamination of samples, both in the field and in the lab, is considerable (Robertson et al. 1987; Gill

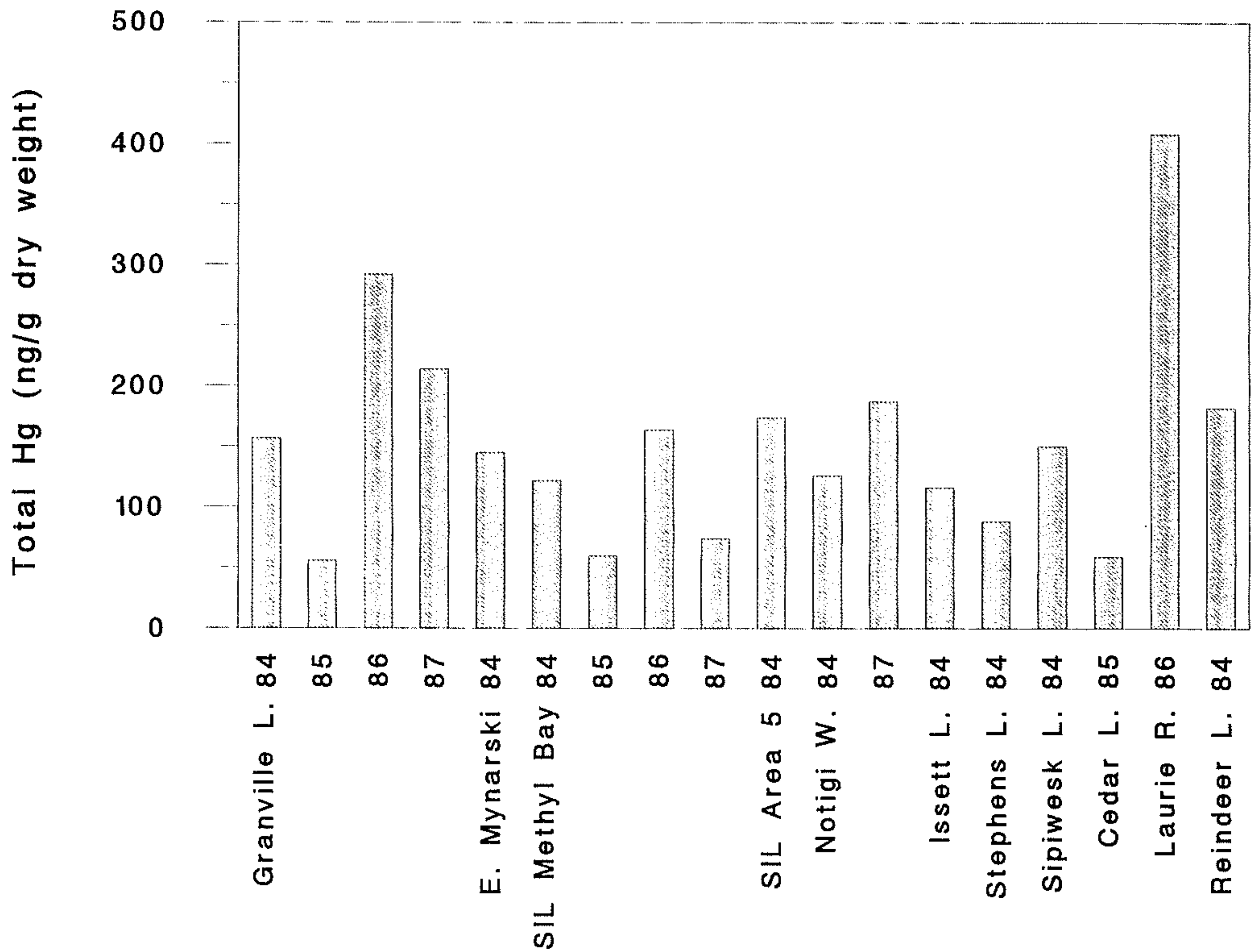

Figure 3. Seasonal mean total mercury concentrations in net-plankton from lakes and reservoirs in northern Manitoba, 1984-1987. Data from Ramsey and Ramlal (1987 a and b), Ramsey (1987a), and Ramsey (1988a).

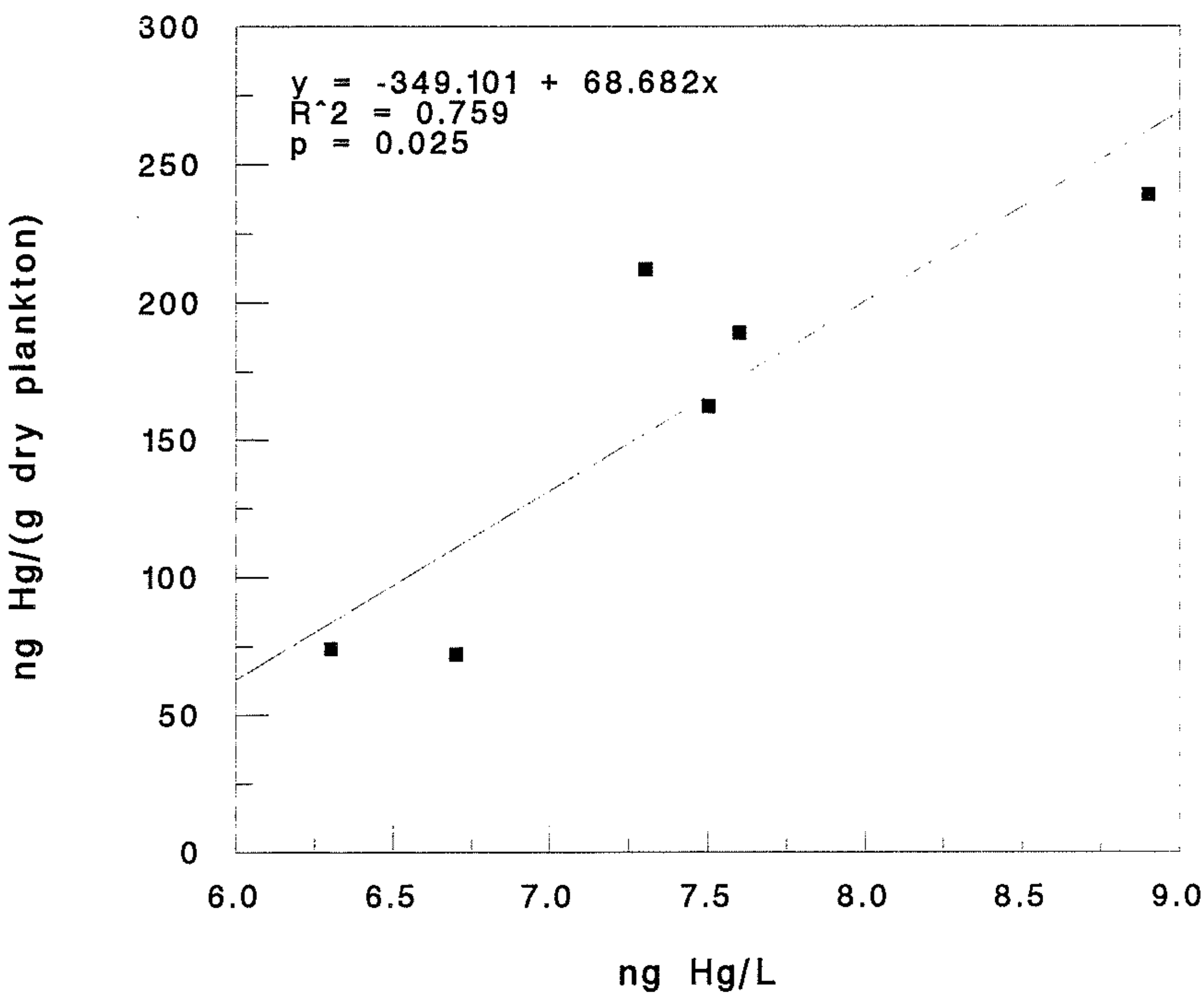

Figure 4. Relationship between seasonal mean total mercury concentrations in water and in net-plankton. From Ramsey (1988a).

Table 2. Total and methyl mercury concentrations in water samples from lakes and reservoirs in northern Manitoba. Data for 1981 from Jackson (1988; Table 10). Data for 1987 and 1989 are from Ramsey (1988a and 1990) and are time-weighted means (with range) for the July-August period.

Year	Lake			Total Hg (ng L^{-1})	Methyl Hg (ng L^{-1})
1981					
	Granville			31.9	0.69 (0.27-1.11)
	Southern Indian				
		Long Bay		23.9	0.29 (0.17-0.40)
		South Bay		26.9	0.30 (0.19-0.41)
	Notigi				
		West		28.3	0.28 (0.38-0.17)
		East	(epilimnion)	21.5	0.60 (0.29-0.91)
		East	(hypolimnion)	36.7	0.59 (0.55-0.63)
	Rat			25.5	0.43 (0.28-0.58)
	Central Mynarski			26.3	0.50 (0.29-0.70)
1987					
	Granville		(offshore)	8.3 (4.5-15.1)	ND
	Southern Indian				
		Methyl Bay	(offshore)	7.2 (5.2-9.0)	ND
	Notigi West		(offshore)	7.4 (4.5-10.3)	ND
1989					
	Granville		(nearshore)	1.44 (0.70-1.82)	0.021 (0.013-0.033)
			(offshore)	1.50 (0.66-2.37)	0.023 (0.020-0.038)
	Southern Indian				
		Methyl Bay	(nearshore)	1.18 (0.71-2.37)	0.034 (0.028-0.045)
			(offshore)	1.60 (0.73-3.15)	0.017 (0.014-0.023)
	Notigi West			1.07	0.046
	Burntwood River at Thompson			2.10	0.027 (0.023-0.031)
	Stephens		(nearshore)	1.77 (1.07-2.70)	0.037 (0.027-0.044)
			(offshore)	1.28 (0.81-1.73)	0.054 (0.041-0.060)

ND No data.

and Fitzgerald 1985). As these sources of contamination are controlled, the measured concentrations decline.

The most important source of contamination appears to have been the sampler used in collection. The highest concentrations (1981 and 1987; Table 2) were measured in samples collected with various types of plastic samplers (i.e., van Dorn bottle or plastic jug, Jackson 1988; van Dorn bottle, Ramsey 1988a), while the low levels measured in 1989 were in samples in which the bottles were filled directly from the lake (Ramsey 1990). Extensive measures also were taken in 1989 to clean the sample bottles before use, to protect the samples from contamination while in the field, and to minimize the mercury content of analytical reagents (Ramsey 1990).

2.2 Methyl Mercury

Although total mercury levels in waters of the CRD have not been elevated as a result of flooding, methyl mercury levels have increased (Ramsey 1990). Until recently, it was necessary to assume that methyl mercury levels were elevated due to the lack of an appropriate analytical method. This assumption was confirmed in 1989 using a newly developed analytical method (Bloom 1989). The methyl mercury level in the flooded zone of Methyl Bay was 55% higher than in Granville Lake while the level outside the flooded zone was just 77% of the Granville Lake value (Table 2). Elevated methyl mercury concentrations also were measured in Notigi Reservoir, near the outflow, and in Stephens Lake on the Nelson River. Levels at these sites were about 34% higher than in the flooded zone of Methyl Bay, probably due to the greater extent of flooding in these reservoirs. Methyl mercury was elevated in the offshore zone of Stephens Lake as well, unlike in Methyl Bay. This was consistent with the occurrence of elevated net microbial methyl mercury production in the offshore zone of Stephens Lake but not in SIL (Ramsey 1990).

Methyl mercury accounts for a very small portion of total mercury in water, in the range of 1-5% (Table 2), so there is considerable room for an increase in methyl mercury before an increase in total mercury can be measured. There are no guidelines for methyl mercury levels in water, but it seems very unlikely that even the highest levels found in the CRD and Nelson River represent a health threat. This is best illustrated by comparison to the amount of mercury consumed from fish. A person eating 500 g (about 1 lb) of whitefish with a total mercury concentration, all as methyl mercury, of 200 ng g^{-1} (i.e., 0.2 $\mu g\ g^{-1}$) consumes 100,000 ng (or 100 μg) of methyl mercury. According to Health and Welfare Canada fish consumption guidelines, unlimited quantities of fish with this mercury level can be eaten. The highest single methyl mercury level measured in water in 1989 was 0.06 ng L^{-1} in Stephens Lake (Table 2). In order to consume the same quantity of methyl mercury directly from the water as in the single meal of whitefish, a person would have to drink approximately 1.7 million litres of water, providing a graphic illustration of the ability of methyl mercury to accumulate in fish tissues.

2.3 Relationship Between Total and Methyl Mercury

A significant positive **correlation** between the concentrations of methyl and total mercury in water from Southern Indian and Granville lakes and Notigi Reservoir was reported in one of the early studies on the CRD (Jackson 1988). This suggested methyl mercury production might be limited by mercury availability. However, no correlation between the concentrations of methyl and total mercury was found in the FEMP studies (Table 2; Ramsey 1990).

The contradictory findings of these studies may be a result of the different sampling and analytical methods employed. Jackson (1988) used the solvent extraction procedure of Uthe et al. (1972) for the analysis of methyl mercury. This method was originally developed for the analysis of methyl mercury in fish tissues and subsequently adapted for analyses in water. A detection limit of 0.1-0.2 ng L^{-1} was stated but no basis for this claim was provided. The analyses also were performed on **centrifuge supernatant** rather than whole water, with the centrifugation step representing a potentially serious source of contamination. In the FEMP study, methyl mercury analyses were performed on samples of whole water collected directly from the lake in teflon sample bottles. The sample bottles were cleaned in hot acid before each trip, and were double bagged for transportation and storage. Plastic gloves were worn whenever the bottles were handled. These measures have been found necessary to prevent sample contamination (N. Bloom, Brooks Rand Ltd., Seattle, WA, pers. comm.).

The presence of contamination in Jackson's samples is indicated by concentrations of methyl mercury in water from Granville Lake (0.25 and 1.03 ng L^{-1} as Hg; Table 2) which are 10-100 times higher than found in natural oligotrophic and mesotrophic lakes elsewhere (<0.004-0.039 ng L^{-1}; Bloom 1989). Granville Lake is a pristine lake on the Churchill River upstream from SIL and unaffected by hydroelectric or other industrial developments.

There also was evidence of contamination in Jackson's total mercury samples. The reported concentration in Granville Lake water (32 ng L^{-1} as Hg; Table 2) is substantially higher than measured in other natural waters using clean sampling and analytical techniques (2.8-4.1 ng L^{-1} as Hg; Robertson et al. 1987), and is comparable to levels reported for sites of industrial pollution. Rudd and Turner (1983) reported total mercury concentrations in the range of 18-35 ng L^{-1}, as Hg, in the highly mercury-contaminated Clay Lake, Ontario. Jackson's analytical results clearly are questionable, raising considerable doubt regarding the reported correlation between total and methyl mercury levels in CRD waters. Such a relationship is inconsistent with present knowledge of the factors regulating methyl mercury production in these lakes (Section 3.0). Rates of methyl mercury production and levels in water and fish are not related to total mercury levels, either in source materials or in the water column.

2.4 Downstream Transport of Methyl Mercury

There are a number of indications that elevated methyl mercury production in the extensively flooded areas of the CRD, such as in Notigi Reservoir, may be affecting fish mercury levels in downstream watercourses as well. Derksen and Green (1987) identified 3 lakes on the CRD (Wapisu, Mynarski, and Wuskwatim) where the increases in fish mercury levels following diversion were greater than expected on the basis of the extent of flooding, and suggested downstream transport of methyl mercury from Notigi Reservoir may be responsible. Similarly, Johnston et al. (1991) found the area of flooded land upstream of a lake or reservoir on the CRD was an important factor in determining fish mercury levels.

The higher than expected fish mercury levels downstream of Notigi Reservoir probably are the combined result of movements of mercury contaminated fish downstream through the control structure at Notigi and the discharge of methyl mercury contaminated water. The relative importance of these pathways remains to be determined.

The 1989 survey of methyl mercury levels in CRD waters included measurements in the Burntwood River at Thompson to look for evidence of downstream transport. The range of concentrations at this station was the same as in Granville Lake (Table 2), situated above the reservoirs on the CRD. Although there is considerable evidence to indicate methyl mercury from Notigi Reservoir is transported as far as Wuskwatim Lake, these measurements suggest the methyl mercury does not travel as far as Thompson, 130 km downstream of the Notigi Dam. The actual limit of downstream transport and the mechanism by which the methyl mercury is lost from the water column remain to be examined.

Evidence of downstream effects also has been found on the Ogoki River diversion in northwestern Ontario. Mercury concentrations in spottail shiners from Ombabika Bay on Lake Nipigon, which receives flows from the Ogoki River diversion via the Little Jackfish River (Figure 5), are significantly higher than in fish from other parts of the lake (Figure 6). The highest level is found at the northern end of the bay, at the mouth of the Little Jackfish River, and levels decrease southward with distance from the mouth of the river. The methylation balance in Ombabika Bay is not elevated, indicating the high mercury levels in fish are a product of the downstream transport of methyl mercury from the upstream Ogoki Reservoir (Ramsey 1987b).

Downstream effects have been documented on the La Grande River below the LG-2 Generating Station, where mercury concentrations in resident salmonids have become seriously elevated as a result of the hydro development. The mean concentrations in resident whitefish are 1.02-1.06 $\mu g\ g^{-1}$, compared with 0.38 $\mu g\ g^{-1}$ in anadromous individuals captured in the river and 0.06-0.2 $\mu g\ g^{-1}$ in other natural rivers draining into James Bay (Roy 1990). Levels in whitefish resident in the river below the LG-2 dam also are much higher than in fish from the LG-2 reservoir, where mercury concentrations were 0.40-0.54 $\mu g\ g^{-1}$ between 1984 and 1988. The

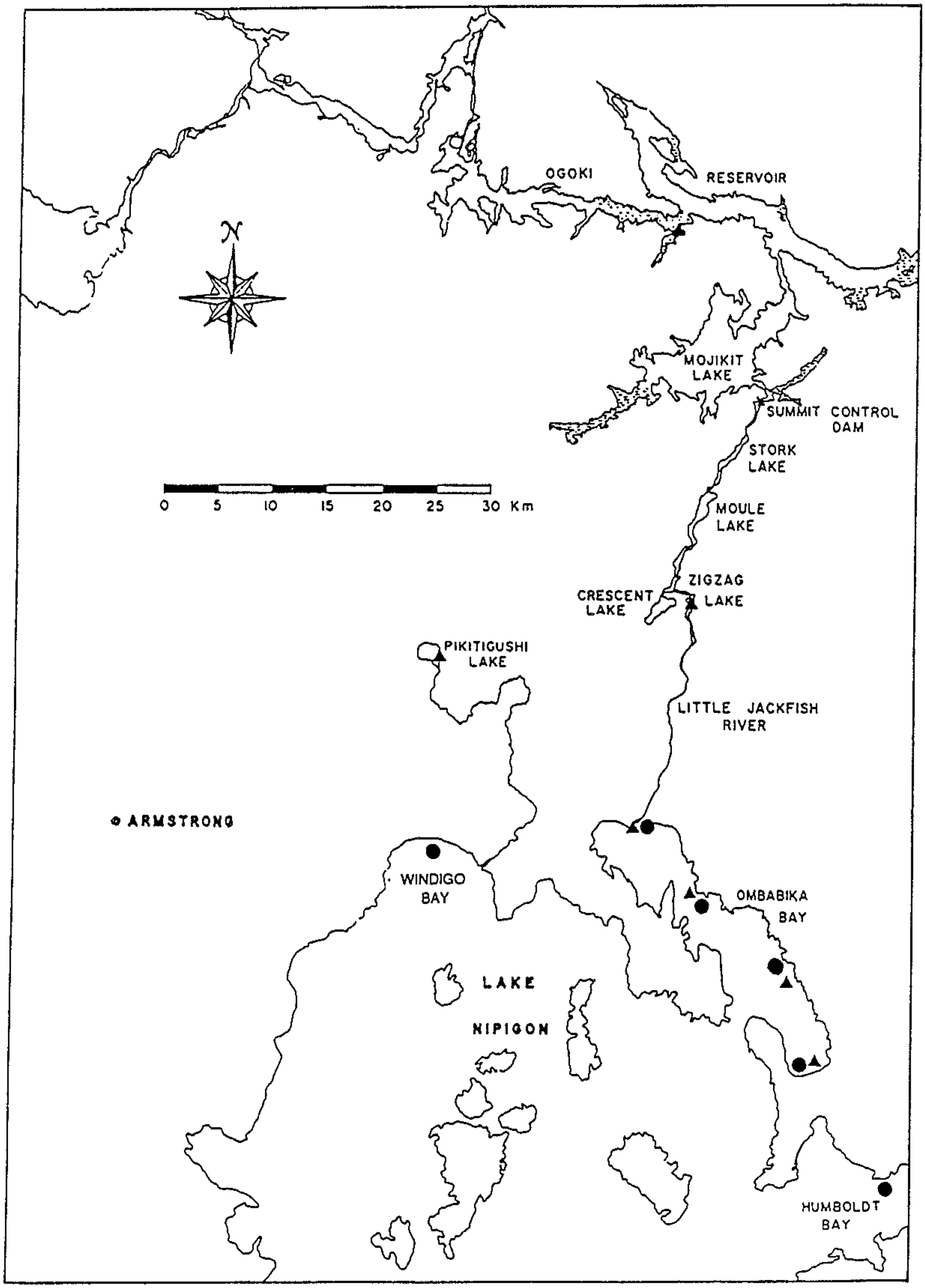

Figure 5. Location of fish mercury (●) and methylation balance (▲) sampling sites on Lake Nipigon and the Little Jackfish River system, 1986.

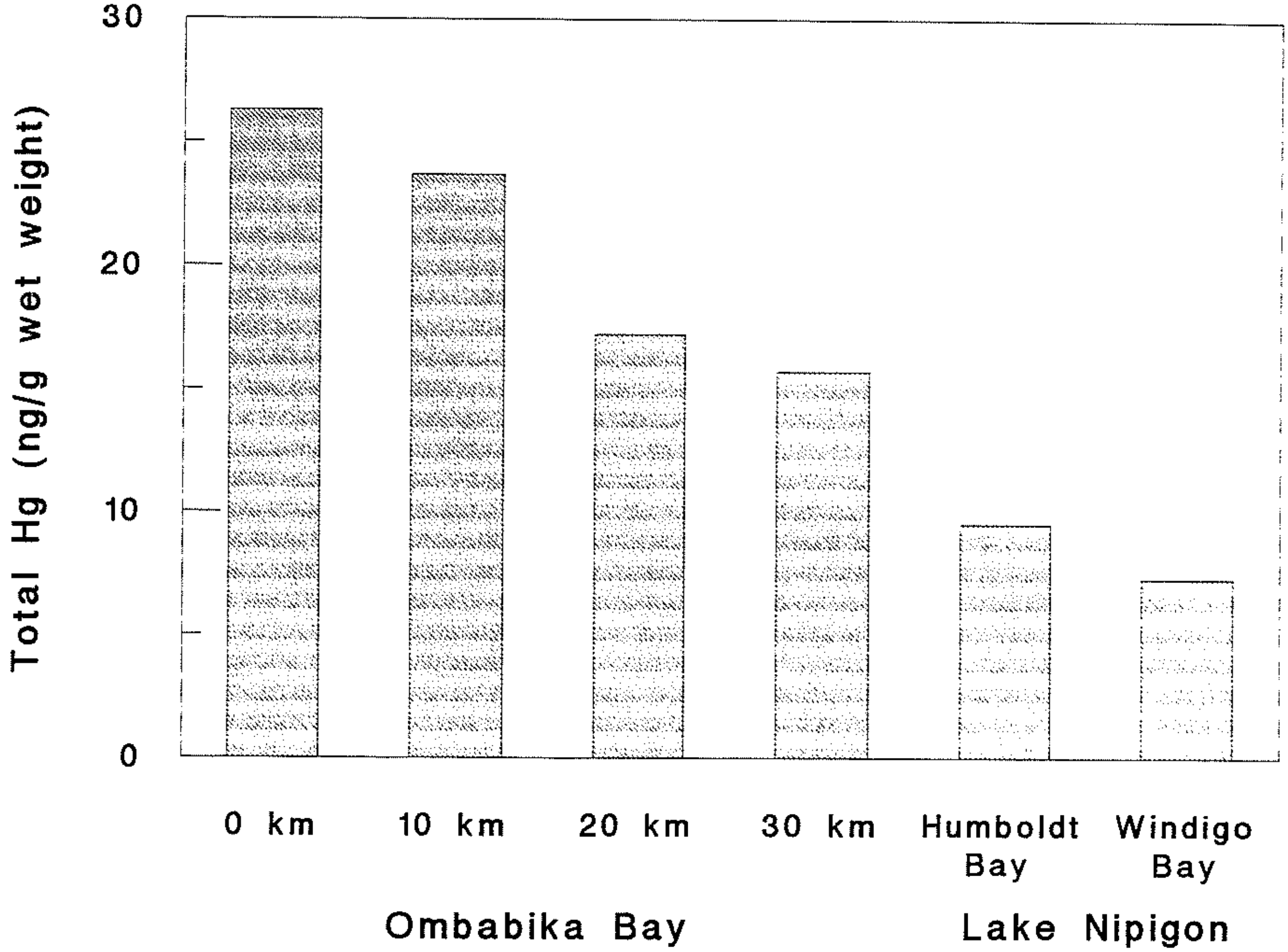

Figure 6. Total mercury concentrations in spottail shiners from sites on Lake Nipigon, August 1986. See Figure 5 for site locations. Ombabika Bay 0 km was at the mouth of the Little Jackfish River. Other stations in Ombabika Bay were the indicated distance from the mouth of the river. Data from Ontario Hydro, Environmental Studies and Assessments Dept., Toronto, Ontario.

higher downstream levels are speculated to be a result of the whitefish adopting a predatory feeding habit, consuming the remains of fish which have passed through the LG-2 turbines. This hypothesis remains to be tested, however.

2.4.1 Split Lake

Located at the confluence of the Nelson and Burntwood rivers, and with reservoirs located upstream on both of these major tributaries, Split Lake (Figure 1) would appear to be a likely site to find elevated mercury levels in fish as result of downstream transport. Mercury concentrations in Split Lake fish also are high enough to suggest a possible upstream influence. On average, the standardized mercury level in walleye captured in survey samples was 0.53 $\mu g\ g^{-1}$ between 1983 and 1989 (Ramsey 1991), and 29% of the walleye catch had a mercury concentration in excess of 0.54 $\mu g\ g^{-1}$ (Green 1990). The mean standardized mercury level in pike was 0.38 $\mu g\ g^{-1}$, and 20% of the survey catch exceeded 0.54 $\mu g\ g^{-1}$ during the same period (Green 1990). Because these levels exceed 0.2 $\mu g\ g^{-1}$, some residents of the Split Lake community with blood mercury levels in excess of 20 ppb have been advised by Health and Welfare Canada to avoid the consumption of pike and walleye as a staple in their diet (B. Stiff, Health and Welfare Canada, Winnipeg, MB, pers. comm.). This restriction has caused concern in the communities and raised questions regarding the effect of past hydro development on fish mercury levels in the lake.

There is no means of directly proving that fish mercury levels in Split Lake are elevated in the absence of pre-development data. The levels in pike and walleye also are not unusual compared to the natural range in northern Manitoba (Derksen and Green 1987). However, the level in walleye in particular appears to be uncharacteristically high for lakes on the Nelson and Burntwood river systems which either have not been flooded, or prior to flooding by hydro developments. The mean mercury levels in survey samples of walleye and northern pike from Playgreen Lake in 1978 and 1981 were 0.20 and 0.21 $\mu g\ g^{-1}$, respectively, and no more than 2% of the catch of either species exceeded 0.54 $\mu g\ g^{-1}$ (Rannie and Punter 1987). The ranges of mean levels in walleye and pike from Cross Lake (west basin) were 0.28-0.33 and 0.31-0.40 $\mu g\ g^{-1}$, respectively, between 1987 and 1989. Less than 6% of the walleye catch exceeded 0.54 $\mu g\ g^{-1}$ (Green 1990). Walleye mercury in Wuskwatim Lake on the Burntwood River averaged 0.36 $\mu g\ g^{-1}$ in commercial samples prior to flooding by the Churchill River diversion (Bodaly et al. 1984).

The seemingly high mercury levels in walleye and pike are not related to Churchill River diversion as they were prevalent before the diversion began operating (Figure 7). The measurements of methyl mercury in water made in 1989 also indicate methyl mercury produced in the Southern Indian Lake and Notigi reservoirs does not travel as far as Thompson and, therefore, could not be affecting fish mercury levels in Split Lake, farther downstream. Mercury concentrations in Split Lake whitefish are negatively correlated

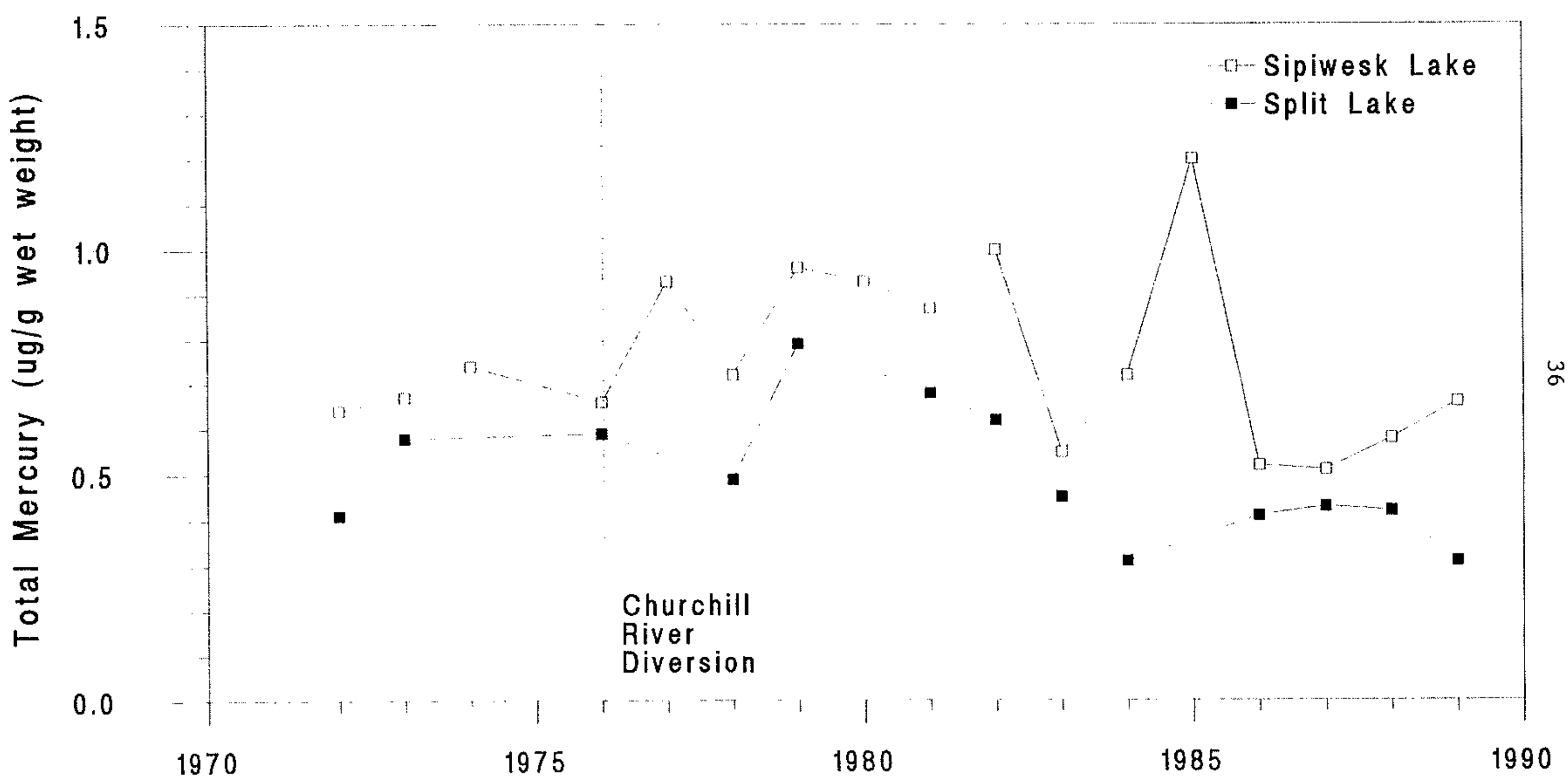

Figure 7. Mean total mercury concentrations in northern pike from Split and Sipiwesk lakes, 1972-1989. Data to 1982 from Derksen and Green (1987). Later data from Green (1990). All data are from lake survey samples.

with Burntwood River discharges (Figure 8), suggesting the waters of the CRD have some form of mitigating influence on methyl mercury availability or uptake in the lake.

It remains to be determined if the methylation balance in Split Lake is naturally high, but this seems to be an unlikely explanation. An external methyl mercury source other than the Nelson River also is improbable. In either case, a negative correlation between mercury levels in whitefish and Nelson River discharge would be expected, because the Nelson River is the major inflow and, therefore, the major means of dilution. However, there is no correlation between mercury in whitefish and Nelson River discharge (Ramsey 1991). This suggests there is a methyl mercury source situated upstream on the Nelson River. Sipiwesk Lake, the Kelsey Dam forebay, and fish killed in the Kelsey Dam turbines are potential sources. In 1987, the microbial mercury methylation balance in the flooded zone of Sipiwesk Lake was elevated compared to the balance in pre-existing lake sediments and a reference lake (Ramsey 1988a). Mercury levels in Split Lake pike also are positively correlated with levels in Sipiwesk Lake pike (Figure 9), further suggesting a direct connection between mercury levels in Split Lake fish and flooding upstream on the Nelson River. The Kelsey forebay has not been sampled but, based on surveys of other reservoirs of equal or greater age, any flooded soils and vegetation probably continue to stimulate mercury methylation there as well. The possibility of methyl mercury becoming more available in Split Lake as a result of fish passing through the turbines of

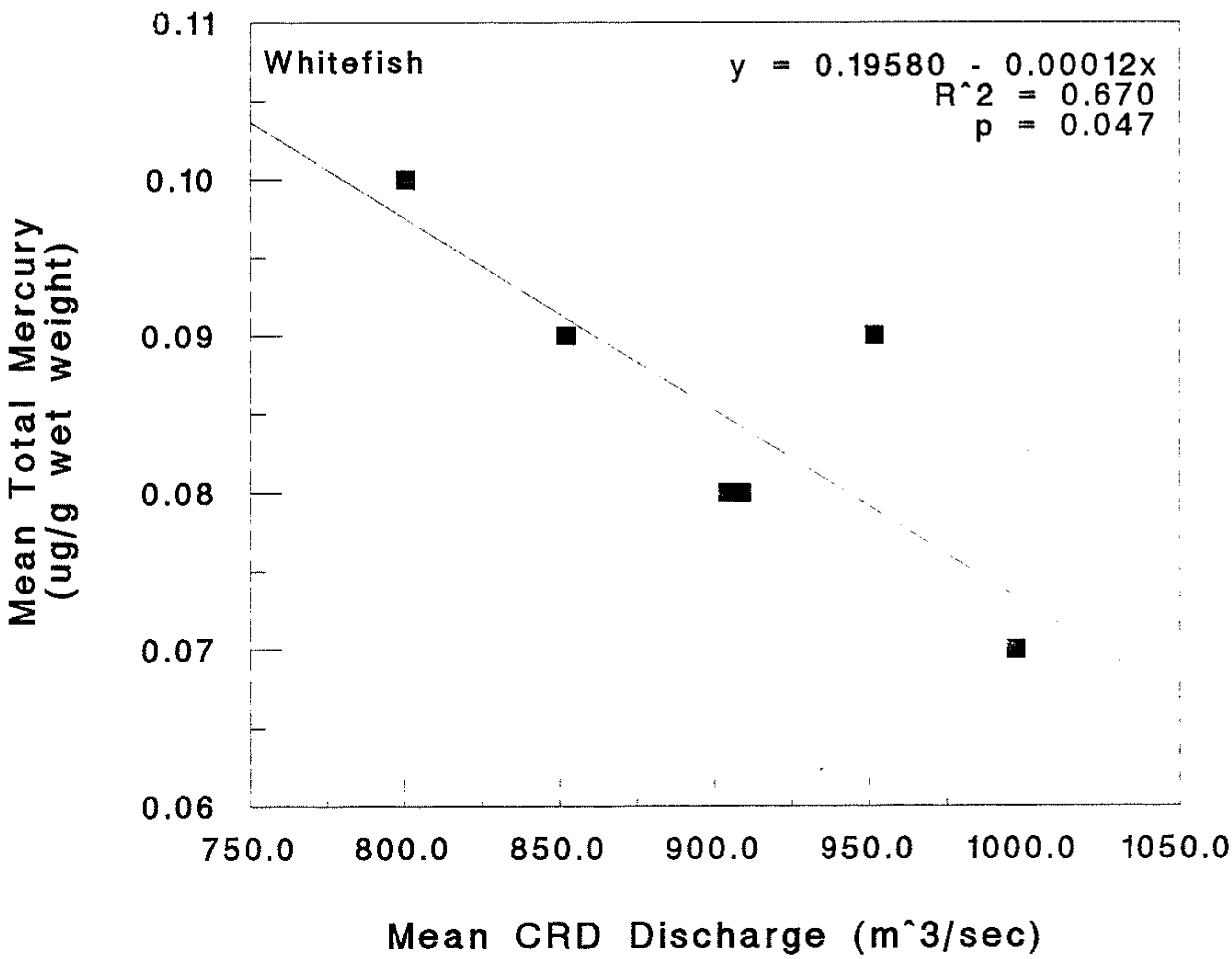

Figure 8. Relationship between length-standardized (350 mm) mercury concentrations in Split Lake whitefish and mean Burntwood River discharge. Discharge data (Water Survey of Canada 1990) averaged over the June to September period in each year. From Ramsey (1991).

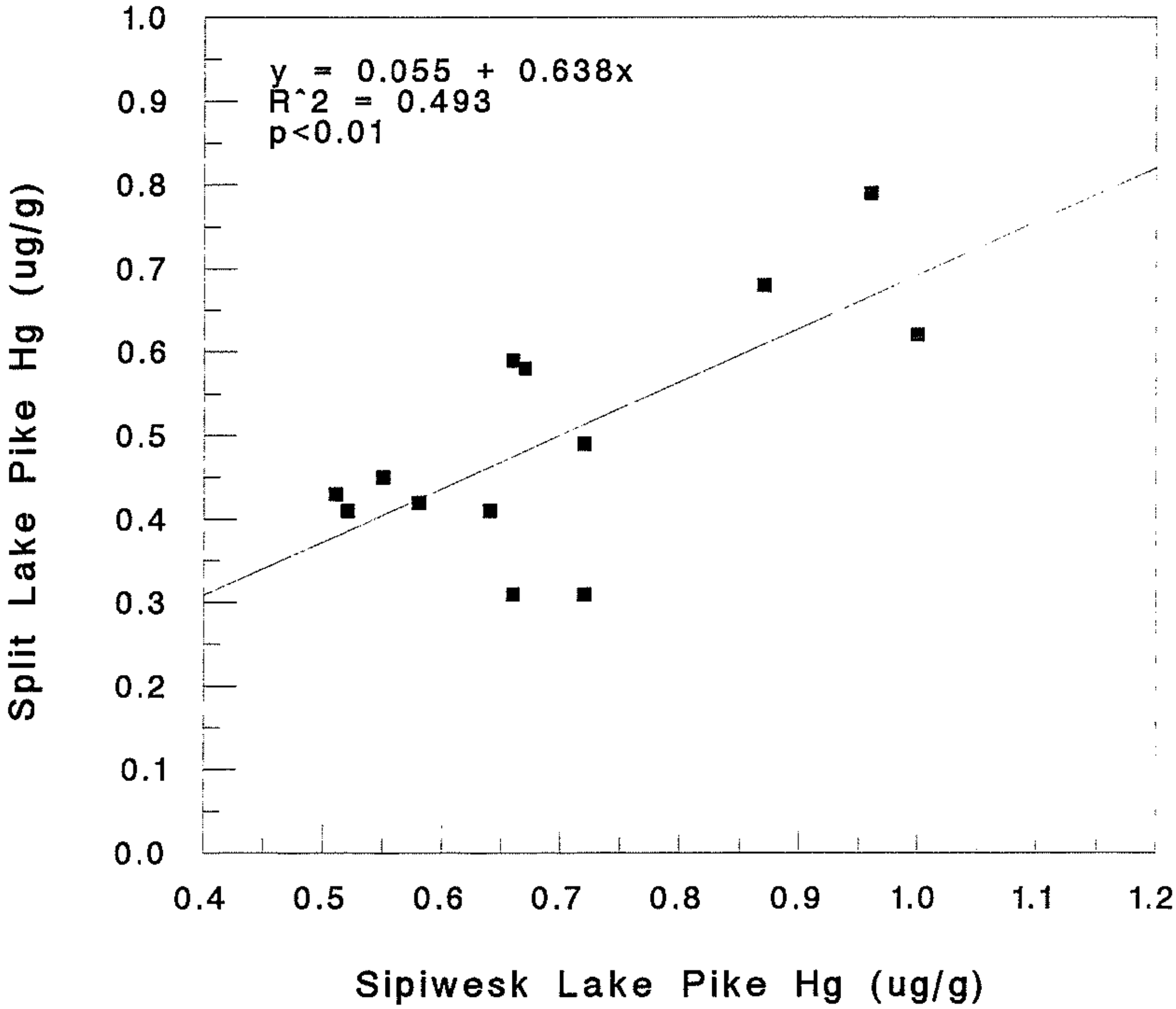

Figure 9. Relationship between mean mercury concentrations in northern pike from Split and Sipiwesk lakes. Data from Figure 7.

the Kelsey Generating Station should be examined as well.

The present significance of hydro developments upstream on the Nelson River as a methyl mercury source to Split Lake could be evaluated using measurements of methyl mercury in water. A series of measurements on the Nelson River between Lake Winnipeg and Split Lake would determine if methyl mercury levels in the river are unusually high and if methyl mercury is being generated at a specific site. The relative importance of other sources (e.g., the Aiken and Burntwood rivers) could be determined in a similar manner. This approach would quantify any continuing contribution of hydroelectric developments on the upper Nelson River to methyl mercury availability in Split Lake.

3.0 Factors Regulating Methylation and Demethylation

3.1 Methylation

3.1.1 Mercury and Organic Matter

The role of the mercury contained in flooded and eroding soils and vegetation in microbial methylation, and its relationship to mercury levels in fish, has been the subject of considerable study on the CRD. This emphasis appears to be a result of the seemingly logical connection between increased mercury loading and the elevation of mercury concentrations in fish. Indeed, this is the approach used for most pollutants. However, it must be kept in mind that the mercury which is accumulating in fish is methyl mercury, while inorganic mercury species predominate in the soils and vegetation. The inorganic mercury in these materials must first be methylated in order for it to move into the fish. The only means by which an increase in inorganic mercury loading could cause an increase in fish mercury levels is if the mercury occurs in a form which is readily available for methylation and if methylation is limited by mercury availability. In this regard, mercury is unlike other pollutants.

There are several indications that microbial methylation, at least in the sediments of lakes and reservoirs in northern Manitoba, is not limited by inorganic mercury availability. Rannie and Punter (1987), using data from 42 lakes in the area of the Churchill River diversion, found there was no relationship between total mercury concentrations in lake sediments and mercury concentrations in northern pike. Bodaly et al. (1987) also found there was no relationship between the mercury concentration in flooded soils from a number of sites on the CRD and mercury concentrations in forage fish (yellow perch, spottail and emerald shiners) from the same areas.

A consistent finding of the methylation balance studies conducted under the FEMP has been a positive correlation between microbial methylation and the concentrations in sediments of both organic carbon and total mercury (Figures 10 and 11). The mercury and organic carbon levels are strongly correlated with each other (i.e., they are collinear) as well (Figure 12). Although the correlation between methylation and sediment mercury could be interpreted as an indication of the importance of mercury in regulating methylation, evidence from the CRD area and elsewhere

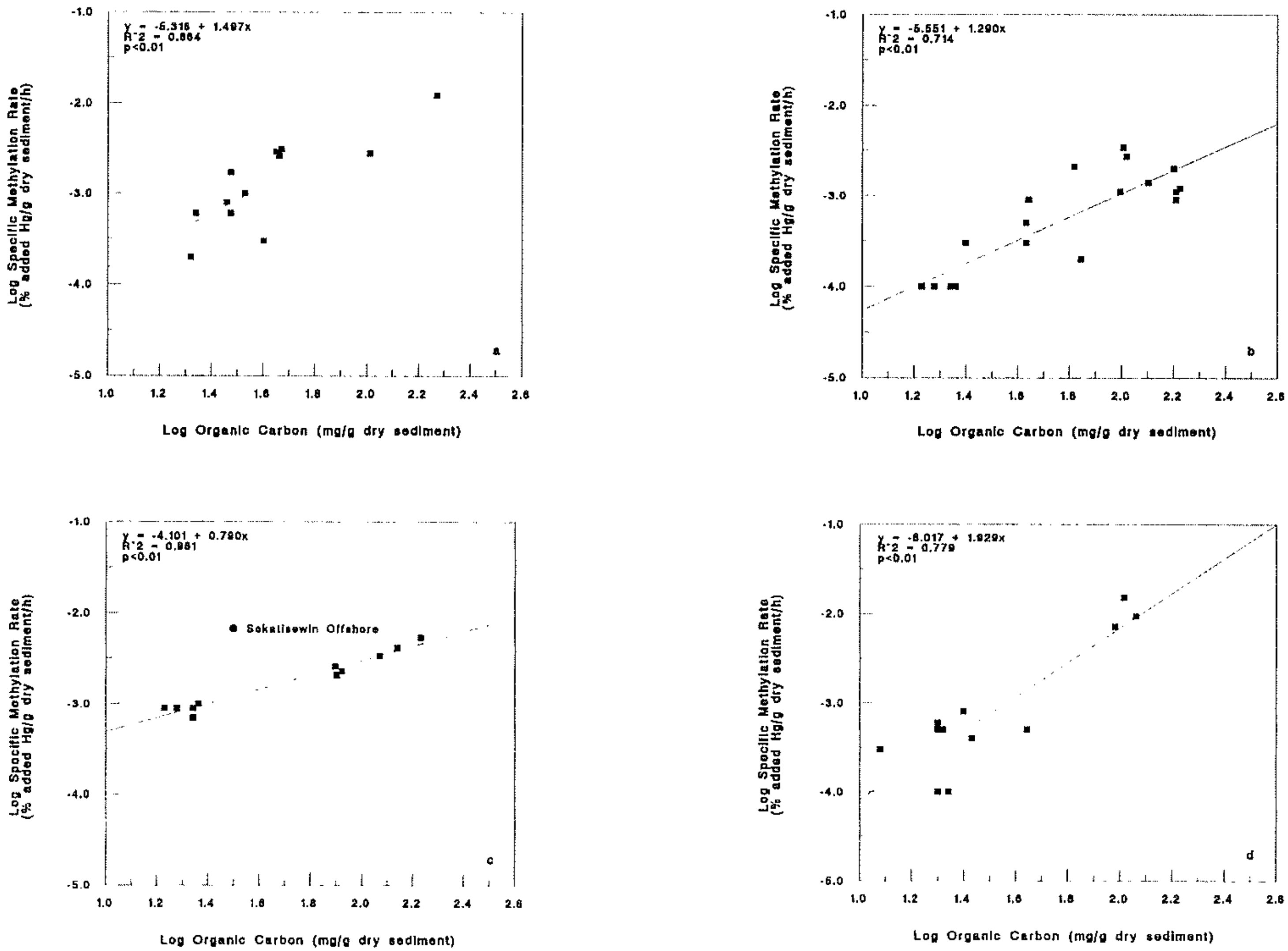

Figure 10. Relationship between seasonal mean organic carbon concentrations and specific methylation rates measured in sediments from lakes and reservoirs in northern Manitoba and Saskatchewan in (a) 1986, (b) 1987, (c) 1988, and (d) 1989. Data for Sokatisewin offshore station excluded from regression in (c). Data from Ramsey (1987a, 1988a, 1989a, and 1990).

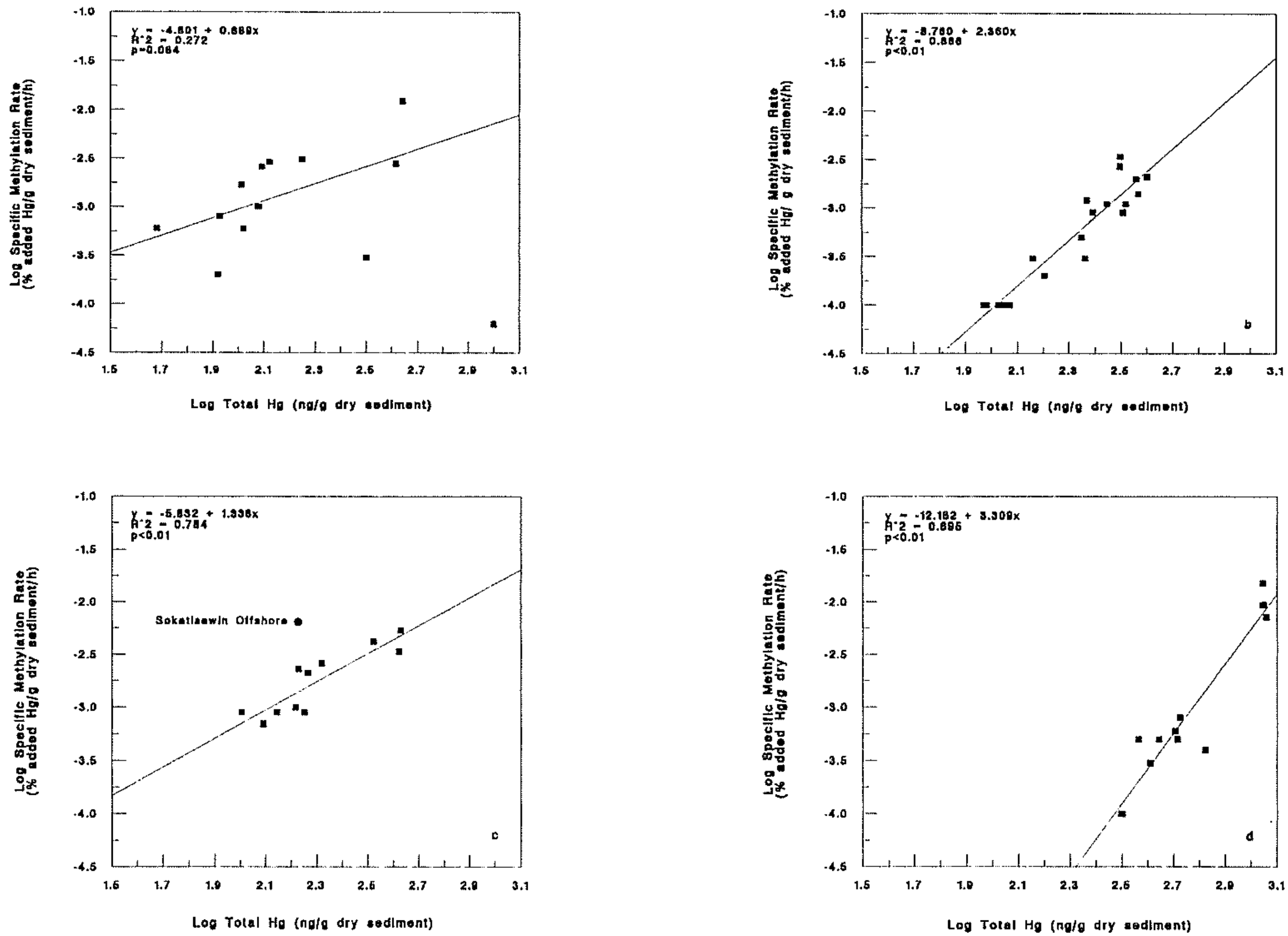

Figure 11. Relationship between seasonal mean total mercury concentrations and specific methylation rates measured in sediments from lakes and reservoirs in northern Manitoba and Saskatchewan in (a) 1986, (b) 1987, (c) 1988, and (d) 1989. Data for Sokatisewin offshore station excluded from regression in (c). Data from Ramsey (1987a, 1988a, 1989a, and 1990).

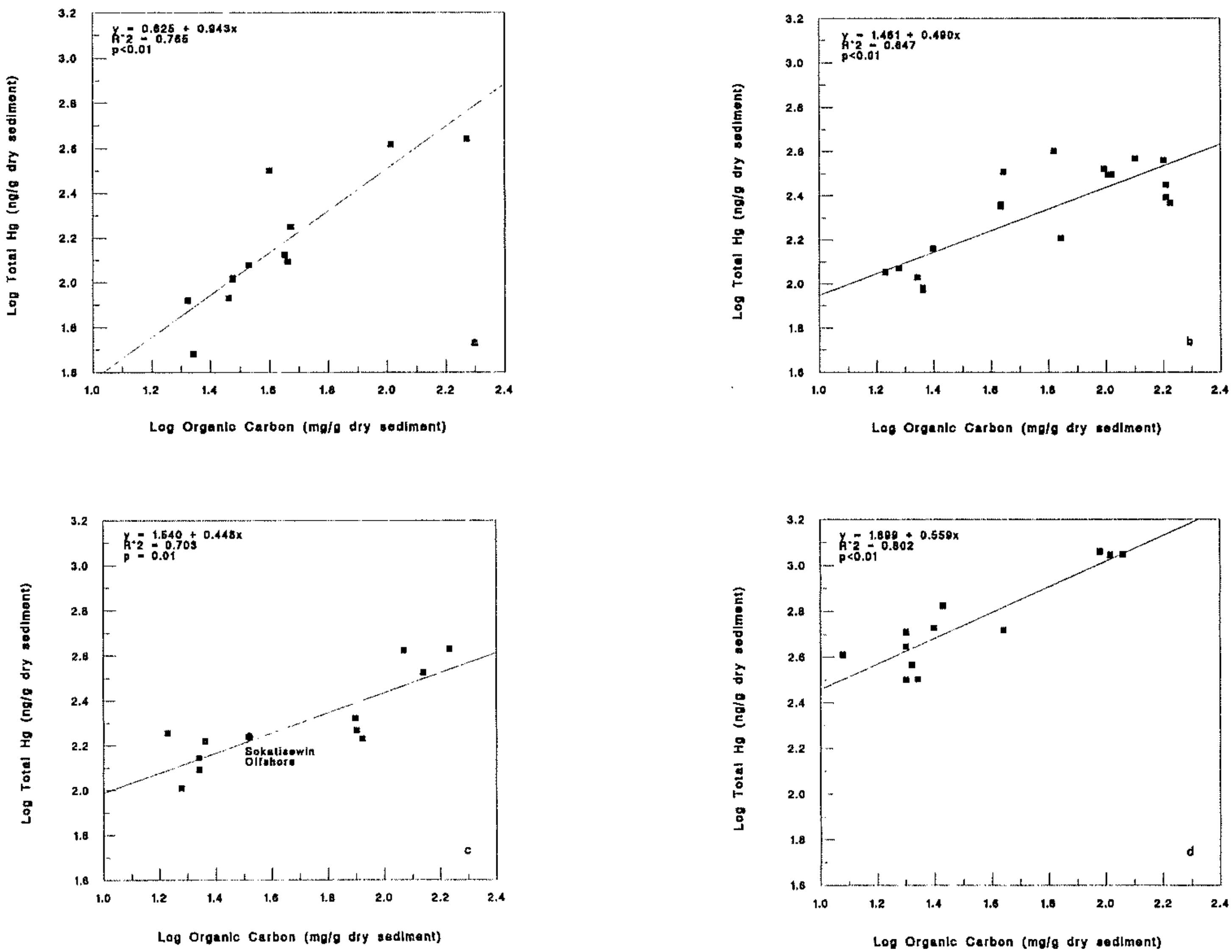

Figure 12. Relationship between seasonal mean organic carbon and total mercury concentrations measured in sediments from lakes and reservoirs in northern Manitoba and Saskatchewan in (a) 1986, (b) 1987, (c) 1988, and (d) 1989. Data from Ramsey (1987a, 1988a, 1989a, and 1990).

suggests the relationship is just a product of the collinearity between mercury and organic carbon. When the correlation between organic carbon and total mercury is removed, either through selection of a subset of the data or sampling in a region where mercury and organic carbon are not correlated, the relationship between mercury and methylation disappears, leaving the correlation between organic carbon and mercury intact. This evidence is detailed below.

In 1986, methylation and demethylation rates were measured at three locations: Methyl Bay on SIL, Laurie Reservoir #2 on the Laurie River, and Granville Lake (Figure 1). Four stations were sampled at each site. A marginally significant correlation was found between specific methylation rates and total mercury in sediments when all sites sampled in 1986 were considered together (Figure 11), but this relationship broke down entirely when the data for Methyl Bay were considered separately. Methylation rates at the four stations in Methyl Bay were positively correlated with organic carbon, but not with total mercury in the absence of a correlation between mercury and organic carbon at the four stations (Figure 13). The significant correlation between methylation and total mercury when all sites were considered together also was strongly influenced by a single station, Methyl Bay station 2 (Figure 11). The correlation disappears when this point is removed from the analysis.

In areas where concentrations of total mercury and organic carbon in sediments vary independently, microbial methylation is

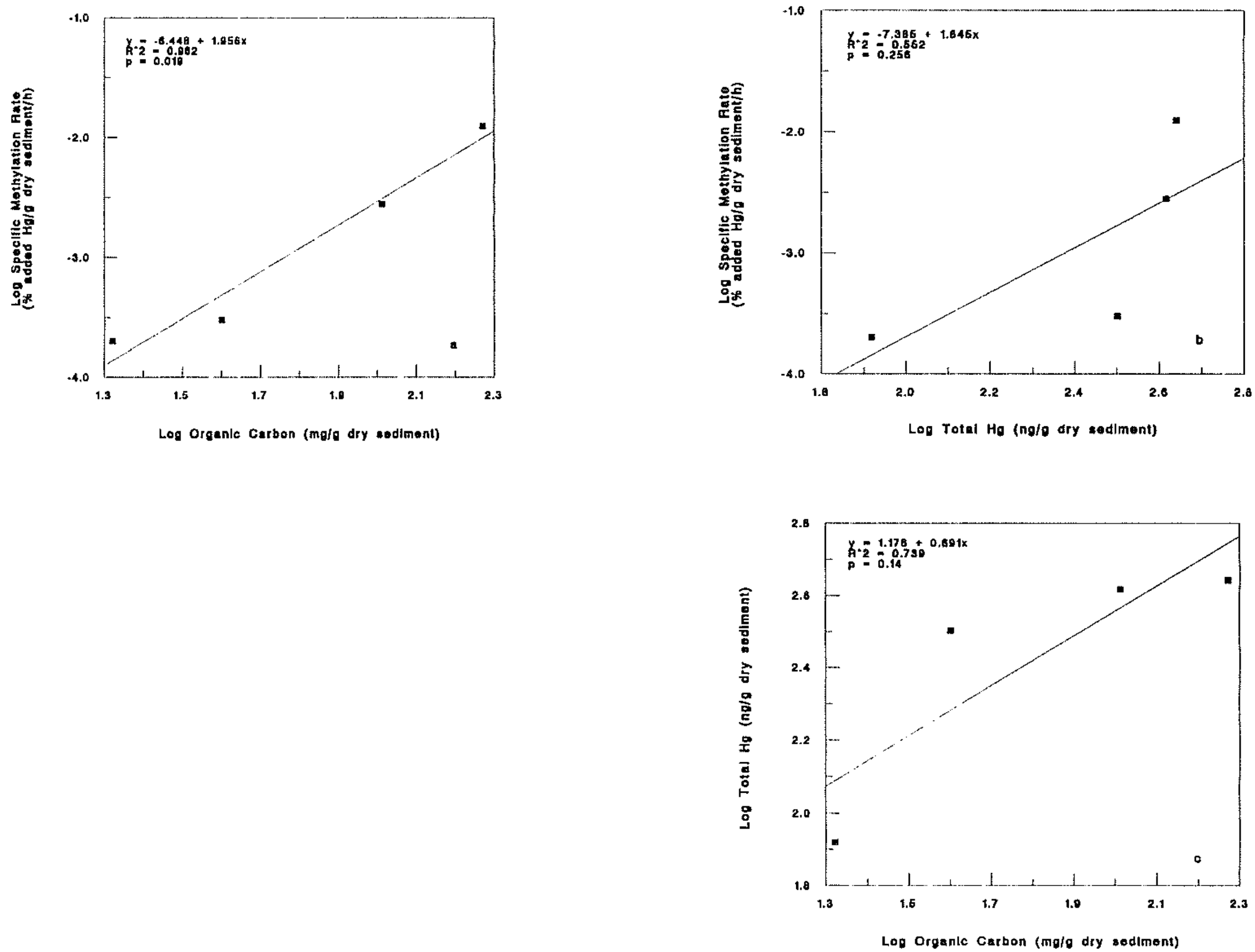

Figure 13. Relationship between (a) seasonal mean total mercury concentrations and specific methylation rates, (b) seasonal mean organic carbon concentrations and specific methylation rates, and (c) seasonal mean organic carbon and total mercury concentrations in sediments from Methyl Bay, Southern Indian Lake, 1986. Data from Ramsey (1987a).

found to be correlated with organic carbon rather than mercury. This is the case in the Little Jackfish River system which flows into Lake Nipigon in northwestern Ontario (Figures 5 and 14), and in streams of the Queen Charlotte Islands (Tables 3 and 4).

Field experiments also have indicated microbial methylation in sediments is not limited by mercury availability. An experiment was conducted on Graham Island, in the Queen Charlotte Islands, British Columbia, in 1988, which examined the influence of soils with a range of total mercury and organic matter contents on microbial methylation and demethylation activity. The experiment involved the submersion of selected soils, contained in plastic basins, in a wetland for approximately 5 weeks before sampling for measurement of specific methylation and demethylation rates using the same radiochemical methods routinely employed in the FEMP studies. Methylation and demethylation rates in the inorganic soils (i.e., soils with less than 20% organic content as measured by **loss on ignition**; Brady 1974) were similar to the rates in Granville Lake in 1988 (Tables 5 and 6). Methylation and demethylation were independent of mercury content, up to 550 ng g^{-1}, the highest concentration in the experiment. Conversely, organic soil (≥20% organic content) with low mercury content (71 ng g^{-1}, 45% lower than in Granville Lake), supported a specific methylation rate comparable to the mean methylation rate measured at the nearshore stations in Methyl Bay in 1988.

The field studies of microbial methylation balance and mercury levels in fish on the CRD indicated that neither was dependent upon

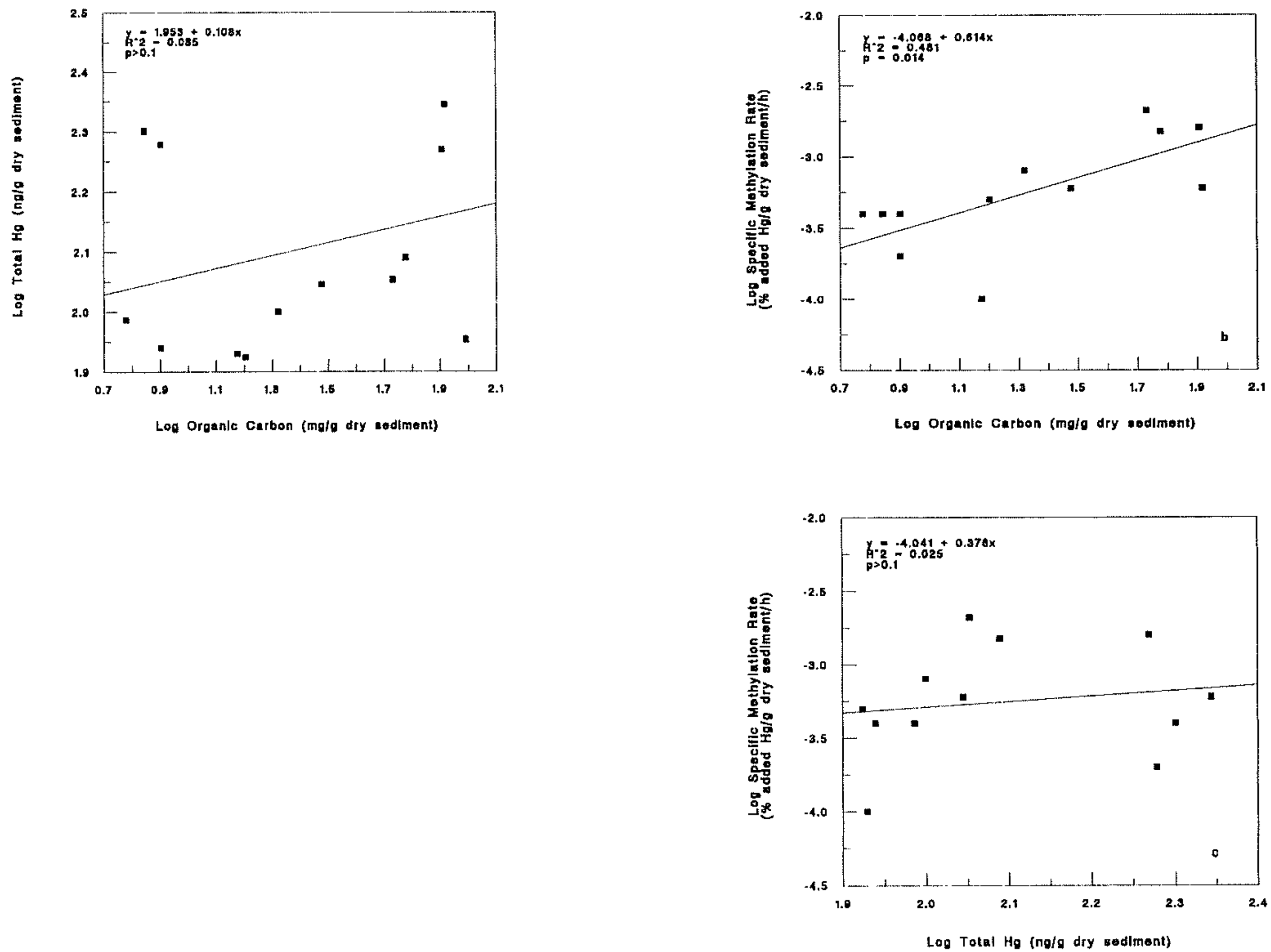

Figure 14. Relationship between (a) seasonal mean organic carbon and total mercury concentrations, (b) seasonal mean organic carbon concentrations and specific methylation rates, and (c) seasonal mean total mercury concentrations and specific methylation rates in sediments from sites on the Little Jackfish River system, northwestern Ontario. Data from Ramsey (1987b).

Table 3. Comparison of specific rates of biological methylation (M), demethylation (D), and methylation balance (M/D) in Graham Island, British Columbia, stream sediments, 1988. Also listed are seasonal mean and peak M, D, and M/D in flooded forest sediments from Southern Indian Lake, Manitoba, and in Granville Lake, Manitoba, an upstream reference lake. Stream data from Ramsey (1988b). Manitoba data from Ramsey (1989a).

Site	Sample	September 20-22		
		M	D	M/D
Middle Barbie Creek	1	<0.001	0.192	<0.005
	2	<0.001	0.362	<0.003
	3	<0.001	0.405	<0.002
	4	<0.001	0.657	<0.002
	Mean	<0.001	0.404	<0.003
Barbie Wetland	1	0.131	0.233	0.562
	2	0.100	0.350	0.286
	3	0.059	0.318	0.186
	4	0.190	0.612	0.310
	Mean	0.120	0.378	0.336
Lower Barbie Creek	1	0.009	0.144	0.063
	2	0.002	0.142	0.014
	3	0.007	0.051	0.137
	4	0.005	0.060	0.083
	Mean	0.006	0.099	0.074
Florence Creek	1	0.001	0.443	0.002
	2	0.001	0.469	0.002
	3	0.003	0.026	0.115
	4	0.002	0.053	0.038
	Mean	0.002	0.248	0.039
Gold Creek	1	0.008	0.026	0.308
	2	0.004	0.177	0.023
	3	0.011	0.083	0.133
	4	0.017	0.098	0.173
	Mean	0.010	0.096	0.159
Southern Indian Lake (July-August 1988)	Peak	0.015	0.540	0.362
	Mean	0.004	0.231	0.017
Granville Lake (July-August 1988)	Peak	0.006	0.394	0.054
	Mean	<0.001	0.058	<0.017

Table 4. Chemical composition of Graham Island stream sediments; September 20-22, 1988. Listed values are the mean and standard deviation (in parenthesis) of analyses on 4 samples. Data from Ramsey (1988b).

Soil Type	Loss on Ignition (% at 550°C)	Total C	Organic C	Total N	Total P	Total Hg
		($mg\ g^{-1}$ dry wt.)				($ng\ g^{-1}$ dry wt.)
Middle Barbie	22.5 (4.9)	89 (25)	85 (25)	3.1 (0.5)	0.7 (0.1)	289 (57)
Barbie Wetland	56.4 (0.4)	275 (10)	269 (10)	14.4 (0.6)	1.5 (0.1)	216 (21)
Lower Barbie	22.2 (5.3)	96 (27)	92 (27)	4.7 (1.1)	1.0 (0.1)	247 (25)
Gold Creek	33.6 (3.2)	172 (12)	166 (13)	9.4 (0.4)	1.2 (0.1)	281 (12)
Upper Florence	26.3 (3.6)	110 (5)	104 (4)	5.0 (0.5)	1.2 (0.1)	165 (3.6)

Table 5. Chemical composition of soils tested for use as substrates in the constructed wetlands and water storage reservoir. Ranges list results of two determinations on each soil-type, otherwise values are a single determination. Data from Ramsey (1988b).

Soil Type	Loss on Ignition (% at 550°C)	Total C (mg g^{-1} dry wt.)	Organic C (mg g^{-1} dry wt.)	Total N (mg g^{-1} dry wt.)	Total P (mg g^{-1} dry wt.)	Total Hg (ng g^{-1} dry wt.)
MSW1	12.0 - 12.6	49	45	1.3	0.4	108 - 125
MSW2-P	17.0 - 17.6	91	86	2.7	0.8	233 - 234
Pit	17.1 - 17.4	113	109	2.7	0.5	466 - 560
Mudstone	25.6 - 28.4	182	180	2.8	0.2	71 - 71
Water Storage	7.9 - 8.2	24	21	0.8	1.0	63 - 69

Table 6. Comparison of specific rates of biological methylation (M), demethylation (D), and methylation balance (M/D) in Graham Island test-soils. Data from Ramsey (1988b).

	MSW1			MSW2-P			Pit			Mudstone			Water Storage		
Sample	M	D	M/D	M	D	M/D	M	D	M/D	M	D	M/D	M	D	M/D
1	<0.001	0.128	<0.008	<0.001	0.009	<0.111	<0.001	0.268	<0.003	0.001	0.098	0.010	<0.001	0.267	<0.004
2	<0.001	0.179	<0.006	<0.001	0.128	<0.009	<0.001	0.025	<0.040	0.002	0.178	0.011	0.001	0.064	0.016
3	<0.001	0.129	<0.008	<0.001	0.008	<0.125	<0.001	0.015	<0.067	0.013	0.046	0.283	<0.001	0.056	<0.018
4	<0.001	0.074	<0.014	<0.001	0.104	<0.010	<0.001	0.027	<0.037	0.009	0.018	0.500	<0.001	0.191	<0.005
Mean	<0.001	0.128	<0.009	<0.001	0.062	<0.064	<0.001	0.112	<0.037	0.006	0.085	0.201	<0.001	0.145	<0.011

the concentration of total mercury in the flooded soils and vegetation, but not why mercury content was unimportant. The mercury in the flooded materials may not be available for methylation, or the supply of available mercury may not be the primary limiting factor of methylation. This question was answered by the limnocorral experiments conducted at SIL between 1981 and 1986 under the CMMA. Hecky et al. (1987a) found that flooded organic matter increased the pool of mercury available for methylation, but the quantity of mercury which accumulated in fish was independent of the size of this pool. This suggested methylation by the natural microbial communities was not limited by inorganic mercury availability. The elevated mercury bioaccumulation by fish in response to the addition of organic soils and vegetation to limnocorrals, and the positive correlation between microbial methylation and sediment organic content in many different environments suggest organic matter availability is the primary limiting factor of methylation in boreal freshwaters.

3.1.2 Microbial Activity and Redox Potential

Organic matter is thought to regulate microbial methylation by controlling microbial activity. The relationships between methylation, demethylation, and microbial activity were examined in the 1988 survey of Methyl Bay, Sokatisewin Lake, and Granville Lake (Ramsey 1989a). Total microbial activity, as indicated by the rate of dissolved inorganic carbon (DIC) production, was measured routinely along with methylation and demethylation.

Contrary to the hypothesis, methylation rates varied independently of microbial activity (Figure 15). Microbial activity and methylation were both elevated in the flooded zone of Methyl Bay, as expected, but elevated methylation rates also occurred in Sokatisewin Lake, where microbial activity was much lower than in Granville Lake.

The explanation for this discrepancy may be found in another characteristic of the Sokatisewin Lake sediments. Oxygen availability in these sediments, as indicated by the oxidation-reduction (redox) potential measured with a platinum electrode, was much lower than in sediments from either Methyl Bay or Granville Lake (Figure 16). Redox potential was lower at the offshore station than nearshore, and the methylation rate was higher offshore. By itself, redox potential was no better predictor of methylation than was microbial activity. However, when the two variables were included in a bivariate regression model, a significant component of the variation in methylation was accounted for ($\log_{10}(M) = 6.975 + 0.998*\log_{10}(MACT) - 2.832*\log_{10}(REDOX)$; $n=12$, $r^2=0.56$, $p=0.025$; where M is the specific methylation rate in % added Hg methylated (g dry sediment)$^{-1}$ h^{-1}, MACT is the rate of DIC production in μmole (g dry sediment)$^{-1}$ h^{-1}, and REDOX is the oxidation-reduction potential in mV).

The importance of redox potential in regulating methylation activity is not surprising given that obligate anaerobic bacteria are responsible for methylation in flooded sediments in Methyl Bay (Section 1.2.4). The activity of these groups should be expected to

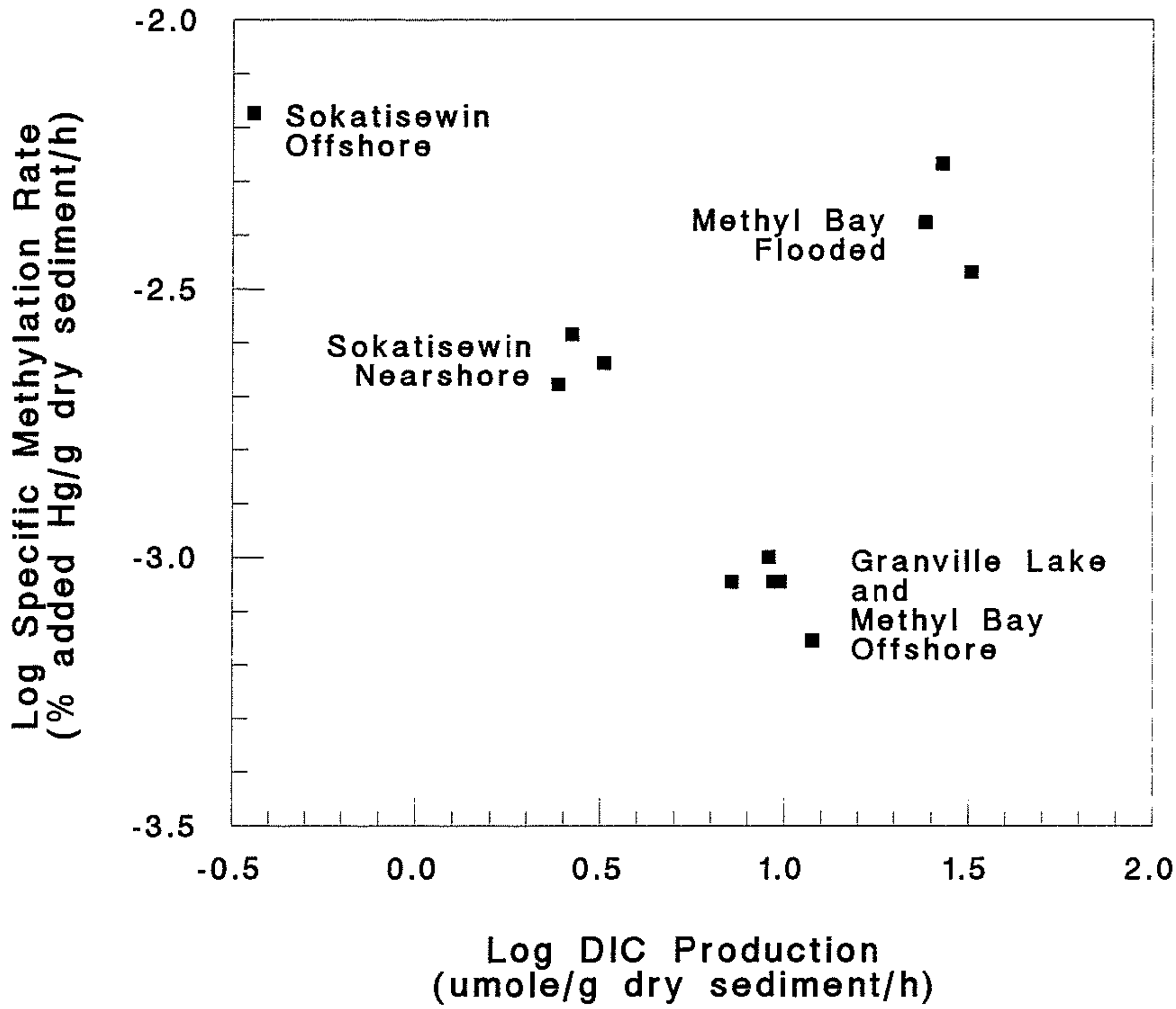

Figure 15. Relationship between the seasonal mean specific methylation rate and microbial activity, as measured by the rate of dissolved inorganic carbon (DIC) production, in sediments from Methyl Bay, Granville Lake, and Sokatisewin Lake, 1988. From Ramsey (1989a).

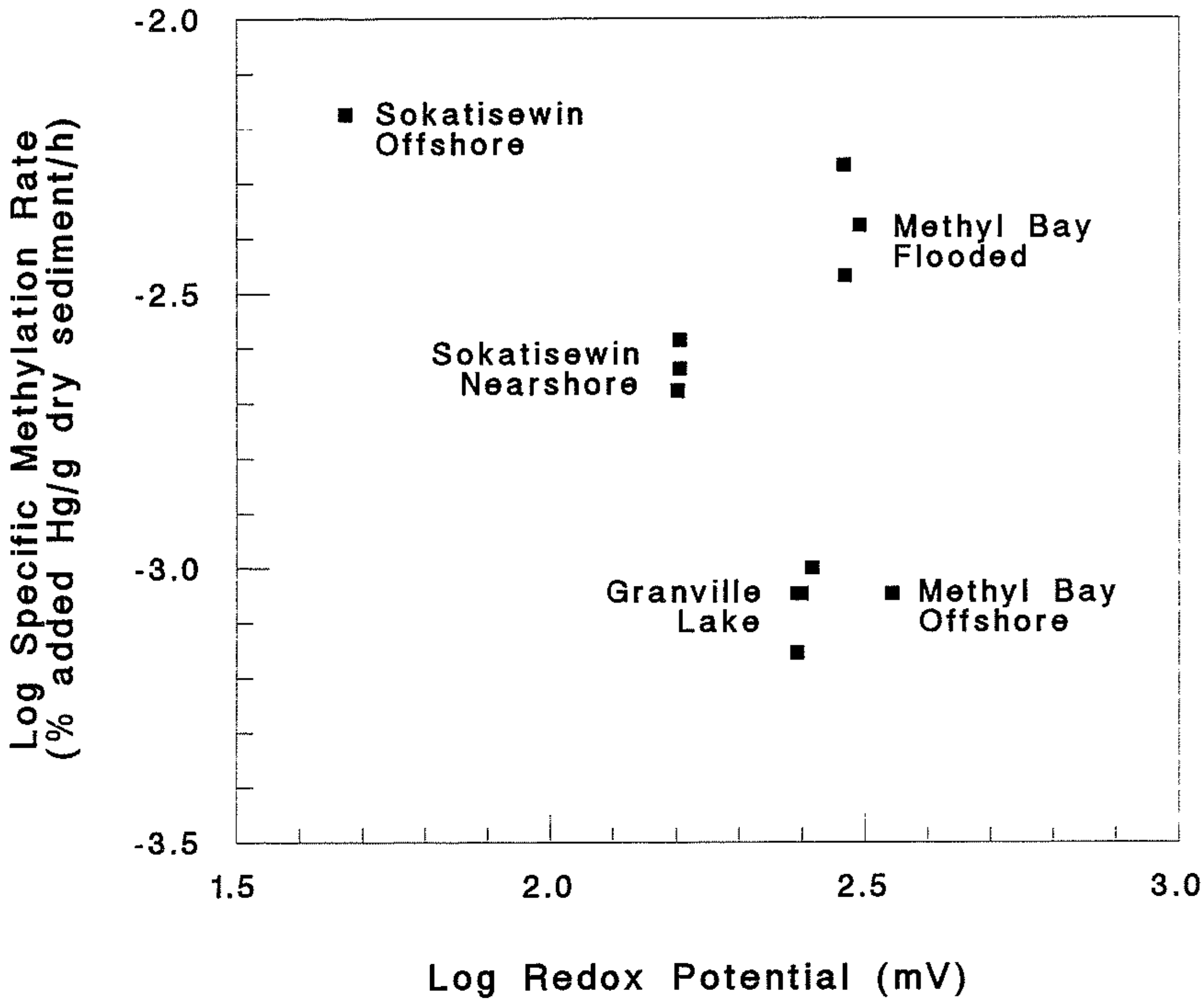

Figure 16. Relationship between the seasonal mean specific methylation rate and redox potential in sediments from Methyl Bay, Granville Lake, and Sokatisewin Lake, 1988. Data from Ramsey (1989a).

increase as oxygen levels decline, provided there is sufficient organic substrate. Although these groups may be more active in reducing conditions, total microbial activity may decrease with decreasing redox potential because the energy yield of the anaerobic process is lower (Mitchell 1974). Thus, the large quantity of organic matter in flooded sediments influences mercury methylation in two ways; providing more bacterial substrate, and promoting the development of conditions which favour the types of bacteria capable of methylation.

3.2 Demethylation

The factors affecting microbial demethylation rates are less well-understood. In the 1986 and 1987 FEMP studies, demethylation rates were positively correlated with methylation rates (Figure 17) suggesting a common limiting factor. Organic matter availability was identified as the controlling variable (Ramsey 1987a and 1988a). However, there was no correlation between demethylation and methylation rates in 1988 nor were demethylation rates correlated with organic carbon (Figures 17 and 18). Instead, demethylation rates were positively correlated with sediment total mercury, microbial activity, and redox potential, and were negatively correlated with sediment pH (Ramsey 1989a). Total mercury was collinear with pH and both were eliminated as regulating variables based on the small range in pH among the stations (6.34-6.80), knowledge of the range of pH which can affect demethylation, and the collinearity of the variables (Ramsey 1989a). Microbial

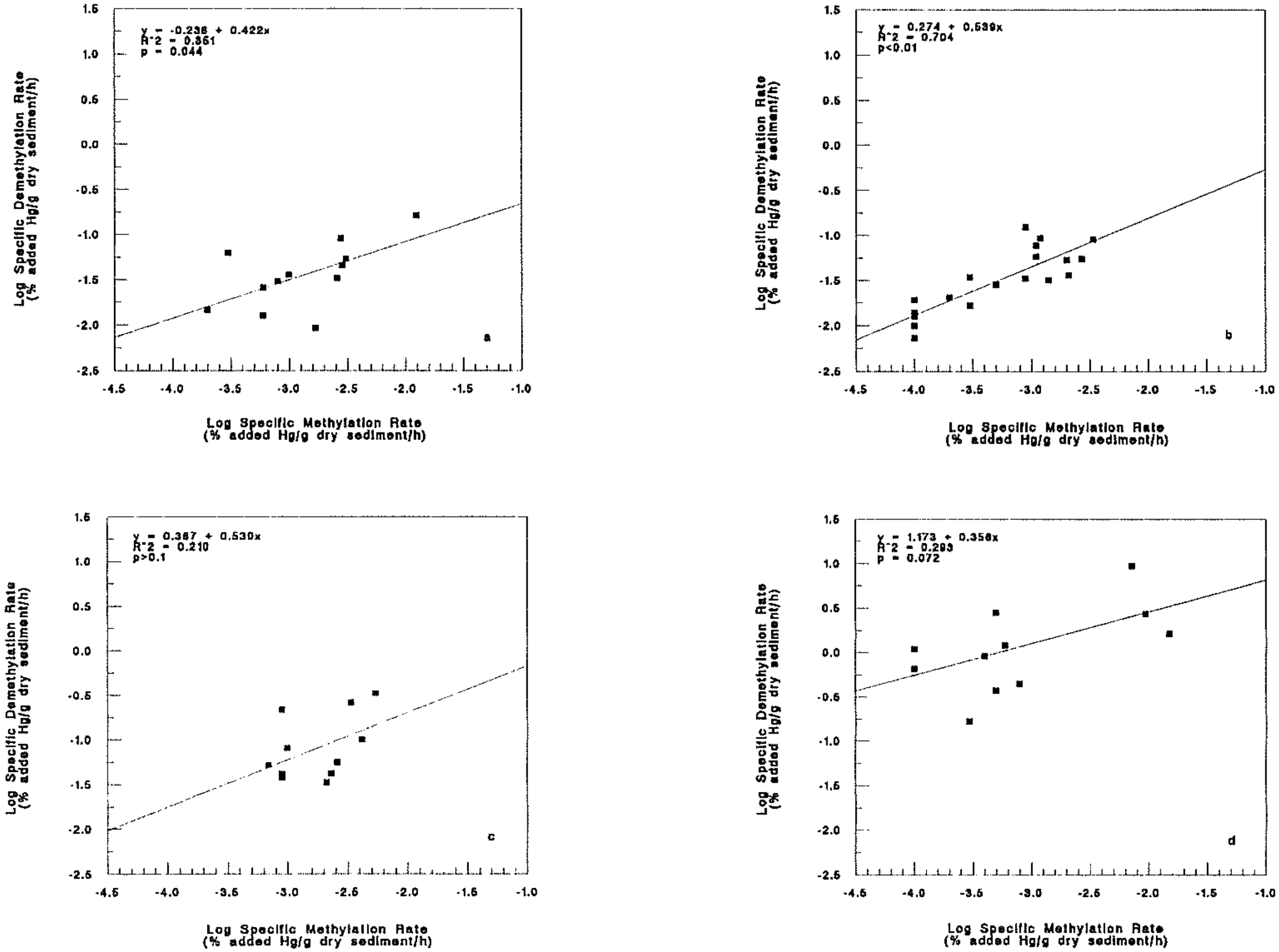

Figure 17. Relationship between seasonal mean specific rates of methylation and demethylation in sediments from lakes and reservoirs in northern Manitoba and Saskatchewan in (a) 1986, (b) 1987, (c) 1988, and (d) 1989. Data from Ramsey (1987a, 1988a, 1989a, and 1990).

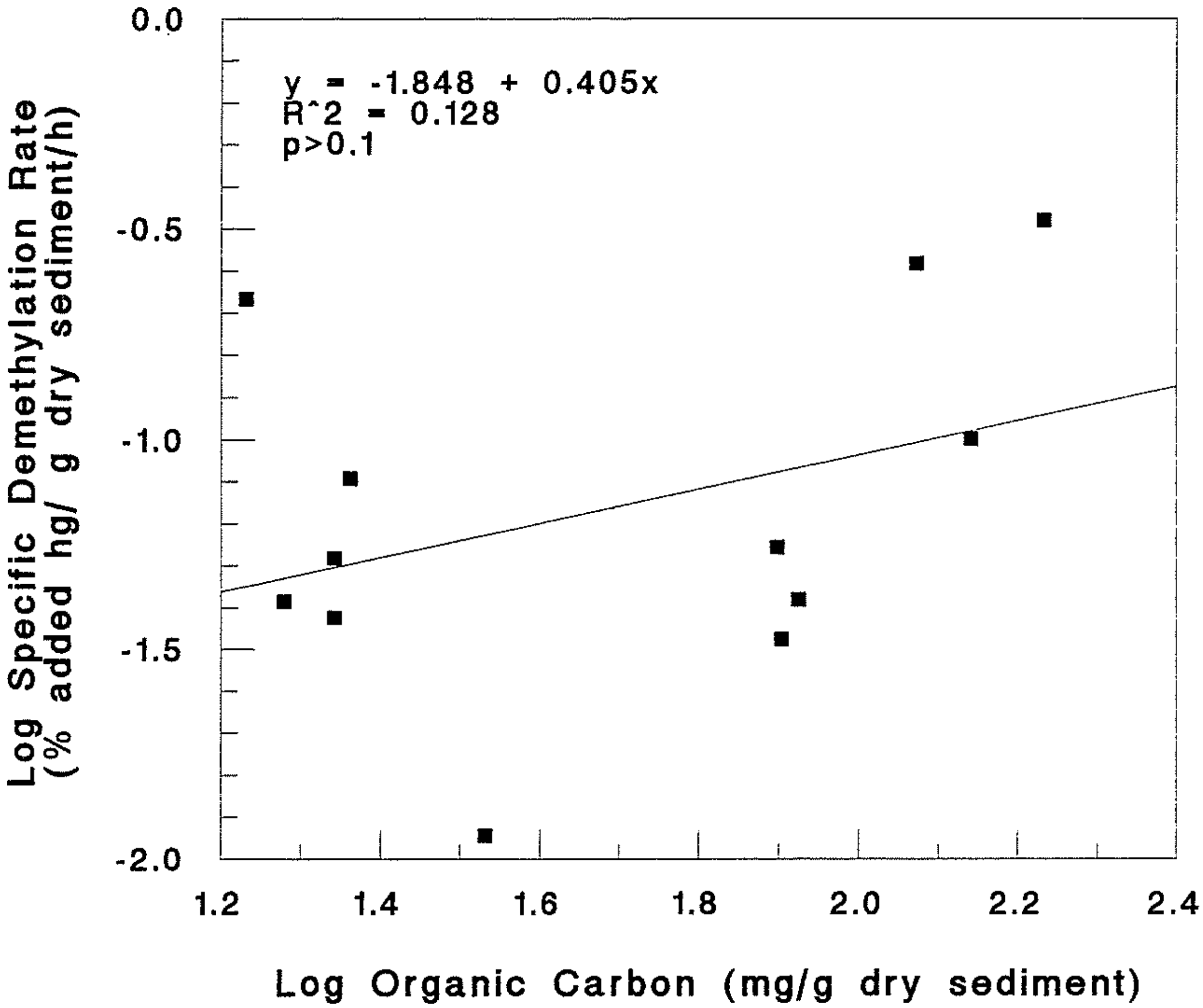

Figure 18. Relationship between seasonal mean specific demethylation rates and organic carbon concentrations in sediments from Methyl Bay, Granville Lake, and Sokatisewin Lake, 1988. Data from Ramsey (1989a).

activity and redox also were collinear, suggesting it was only one of these factors which was responsible. There was no basis for differentiating between microbial activity and redox potential as the regulating factor, nor was there any obvious mechanism by which either could control demethylation (Ramsey 1989a).

A marginally significant correlation between methylation and demethylation rates was found in the 1989 survey of Methyl Bay, Stephens Lake, and Granville Lake (Figure 17). Demethylation also was positively correlated with sediment organic carbon, total nitrogen, and redox potential, and negatively correlated with sediment pH (Ramsey 1990). Again, pH was ruled out on the basis of other studies which examined the effect of pH on demethylation. Redox potential was ruled out as well. Although the differences in redox potential among the sites were highly significant, they were very small in absolute terms (range 149-168 mV). This left organic carbon and total nitrogen which, as was the case in the previous FEMP surveys (Ramsey 1987a, 1988a, and 1989a), also were intercorrelated. This importance of organic matter suggested total microbial activity was the controlling factor but this could not be confirmed directly because the rate of DIC production could not be measured in Stephens Lake due to high carbonate concentrations in the sediments which interfered with the analysis (Ramsey 1990).

An important distinguishing factor between methylation and demethylation is the lack of an influence of redox potential on demethylation rates. This may provide a partial explanation for the differential stimulation of methylation over demethylation in flooded sediments although additional study on the role of sediment oxygen conditions in regulating demethylation is required.

4.0 Duration of the Fish Mercury Problem

When the mercury problem on the Churchill River diversion was first identified in 1979, such problems were thought to be short-lived, with a duration of about 5 years. This projection was found to be incorrect, at least for reservoirs in northern Canada. There was no indication of a decline having started by 1981, five years after the impoundment of SIL and 7 years after Notigi Reservoir began filling. One of the main objectives of the FEMP, therefore, has been to develop and refine projections of the duration of the mercury problem on the Churchill River diversion. This has been approached by:

- monitoring microbial rates of mercury methylation and demethylation at flooded and reference sites in the CRD area;
- monitoring mercury levels in fish from flooded lakes in northern Manitoba; and
- measuring methylation and demethylation rates in older reservoirs located in the northern boreal forest zone.

The Department of Fisheries and Oceans was responsible for the monitoring of fish mercury levels in Southern Indian and Issett lakes. A detailed report of this study is presented by Strange et al. (1991). Monitoring of fish mercury on the CRD and in lakes and reservoirs on the Nelson River was the responsibility of the Province of Manitoba, Department of Natural Resources, Fisheries Branch. Green (1990) documents the data from this study and Ramsey (1991) presents an analysis of temporal trends between 1983 and 1989. The methylation and demethylation studies were undertaken by Agassiz North Associates Limited for Environment Canada, and are reported in Ramsey (1987a, 1988a, 1989a, and 1990).

4.1 Methylation on the CRD

Monitoring of microbial methylation and demethylation on the CRD has concentrated on two sites, a small bay on Southern Indian Lake, called Methyl Bay, representative of flooded conditions, and a reference site on Granville Lake near Leaf Rapids, which has not been affected by flooding (Figure 1).

Specific methylation rates and M/D at the flooded stations in Methyl Bay have been significantly higher than in the nearshore zone of Granville Lake in each of the six years of sampling (Table 7). Specific demethylation rates in Methyl Bay also significantly exceeded rates in Granville Lake in all years but 1984. Not only does the methylation balance remain shifted in favour of methyl mercury production, but there is no evidence of a sustained downward trend in the stimulatory affect of the flooded materials. No consistent declines have occurred in any of methylation, demethylation, or M/D. There was the suggestion of a decline in methylation rates in flooded sediments relative to the reference

Table 7. Comparison of seasonal mean specific methylation and demethylation rates (% added Hg methylated or demethylated (g dry sediment)$^{-1}$ h^{-1}) and corresponding methylation balance (M/D) in the flooded zone of Methyl Bay and in the nearshore zone of Granville Lake: 1984-1989. Values in parentheses are standard deviations of the mean for years when two (1984) or three (1985-1989) stations were sampled. Where no standard deviation is listed, only one station was sampled. Data from Ramsey and Ramlal (1987a and b) and Ramsey (1987a, 1988a, 1989a, and 1990).

	1984	1985	1986	1987	1988	1989
Methylation						
Methyl Bay	0.034 (0.028)	0.022 (0.012)	0.012	0.002 (0.001)	0.0043 (0.0010)	0.0106 (0.0041)
Granville Lake	0.012	0.006 (0.005)	0.0008 (0.0002)	0.0002 (0.0001)	0.0009 (0.0002)	0.0005 (0.0001)
Ratio	2.8	3.7	15.0	10.0	4.8	21.2
Demethylation						
Methyl Bay	0.089 (0.040)	1.200 (0.133)	0.162	0.0468 (0.0127)	0.2312 (0.1185)	4.6102 (4.2504)
Granville Lake	0.162	0.669 (0.216)	0.031 (0.005)	0.0166 (0.0027)	0.0581 (0.0205)	2.2640 (0.9180)
Ratio	0.5	1.8	5.2	2.8	4.0	2.0
M/D						
Methyl Bay	0.383 (0.176)	0.018 (0.012)	0.077	0.0434 (0.0057)	0.0238 (0.0158)	0.0045 (0.0043)
Granville Lake	0.076	0.009 (0.009)	0.026 (0.003)	0.0102 (0.0069)	0.0159 (0.0159)	0.0003 (0.0002)
Ratio	5.0	2.0	3.0	4.3	1.5	15.0

site in 1987 and 1988, but this decline did not continue into 1989, when the largest differential in the 6 year study was recorded (Table 7).

4.2 Mercury in Fish from SIL, the CRD, and Nelson River

Mercury levels in northern pike, walleye, and whitefish in Rat, Threepoint, and Stephens lakes (Figure 1) were surveyed by Manitoba Natural Resources annually between 1983 and 1989. Elevated mercury levels developed in fish from Rat and Threepoint lakes after flooding by the Churchill River diversion between 1974 and 1976, and in Stephens Lake due to flooding by the Kettle Generating Station between 1971 and 1972. Mercury levels have peaked in all species in all three lakes, and appear to be starting to decline. This decline is expected to be slow, however, as indicated by mercury levels in Stephens Lake walleye and pike which remain 3 and 4 times higher than pre-development values 18 years after impoundment. The results of the Fisheries Branch study are summarized below for each lake and are detailed in Green (1990) and Ramsey (1991).

4.2.1 Rat Lake

Rat Lake was flooded between 1974 and 1976 as Notigi Reservoir was created in the Rat River valley (Figure 1). Lake level rose about 10 m (Newbury et al. 1984) and lake area increased approximately 248% (G. McCullough, Department of Fisheries and Oceans, Winnipeg, MB. pers. comm.).

The whitefish mercury data record for the lake starts in 1980, with the peak level, 0.25 $\mu g\ g^{-1}$, occurring in 1984 (Figure 19). Whitefish mercury was stable between 1986 and 1989 at 0.2 $\mu g\ g^{-1}$. Thus, 14 years after the start of impoundment, whitefish mercury remains more than a factor of 2 higher than the average for northern Manitoba lakes remote from hydro development (<0.1 $\mu g\ g^{-1}$; Derksen and Green 1987) and 2.5 to 4 times the pre-development levels upstream in SIL (0.05-0.07 $\mu g\ g^{-1}$; Bodaly et al. 1984) and downstream in Wuskwatim Lake (0.08 $\mu g\ g^{-1}$ in 1970; Bodaly et al. 1984).

The walleye mercury data record for Rat Lake starts in 1978, with uniformly high levels occurring between 1978 and 1984 (Figure 19). Although a decline appears to have occurred between 1978 and 1983, its significance cannot be tested as the pre- and post-1983 data are not directly comparable. The pre-1983 data were not standardized to length in the same manner as the later data. The early samples also were biased to the larger (300+ mm) individuals in the population (Green 1986), which probably explains the apparently higher mean levels during this period.

There was a significant decline in walleye mercury between the 1984 and 1985 samples and again between 1988 and 1989 (Figure 19). Nevertheless, 14 years after impoundment, walleye mercury in Rat Lake remained more than 3 times higher than both the average for northern Manitoba lakes remote from hydro development (0.3-0.4 $\mu g\ g^{-1}$; Derksen and Green 1987) and the pre-development levels upstream in SIL (1971-1975, 0.19-0.30 $\mu g\ g^{-1}$; Bodaly et al. 1984) and

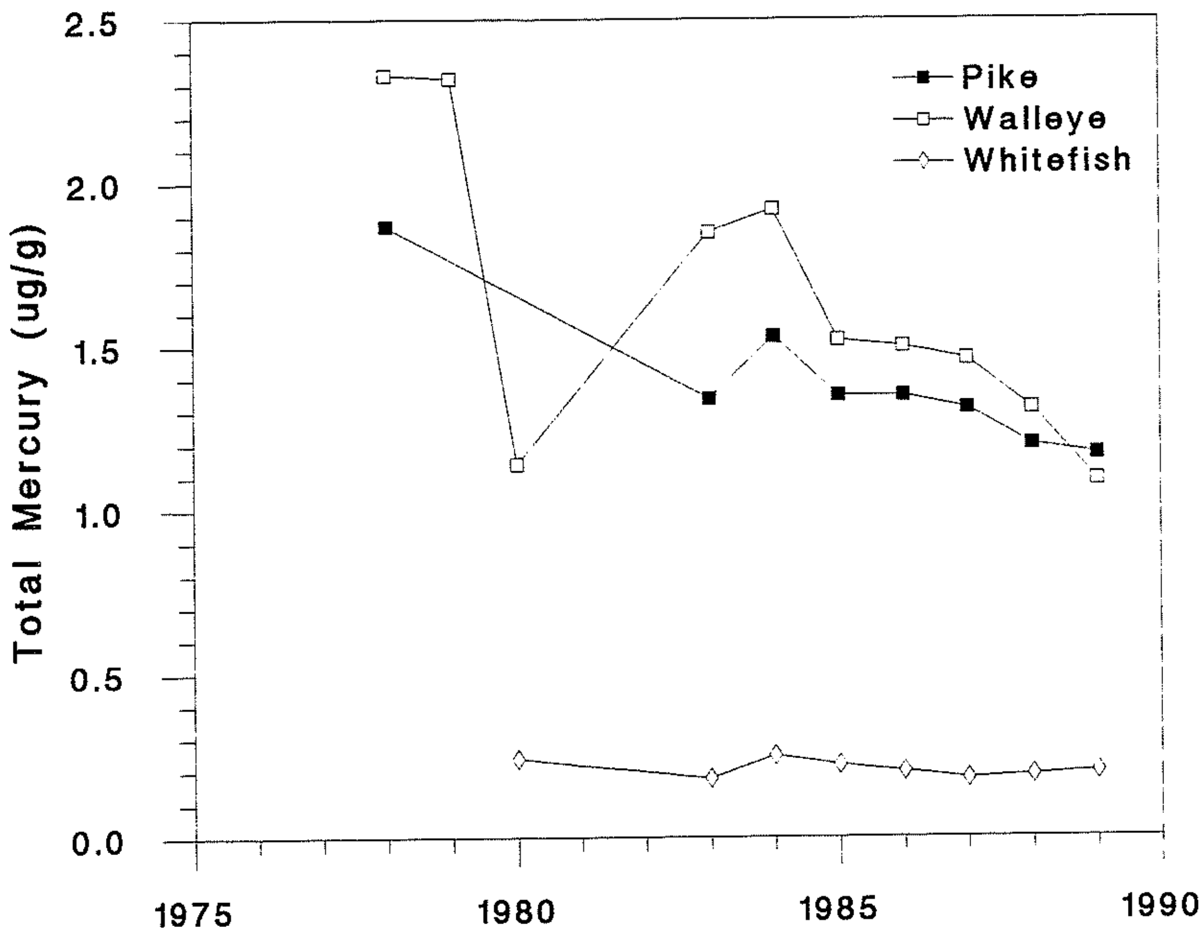

Figure 19. Mean total mercury concentrations in northern pike, walleye, and whitefish from Rat Lake, 1978-1989. Pre-1983 data from Green (1986). Later data are length-standardized means from Green (1990).

downstream in Wuskwatim Lake (1970-1975, 0.25-0.40 $\mu g\ g^{-1}$; Bodaly et al. 1984).

The northern pike mercury data record also starts in 1978 (Figure 19). Again, the significance of any decline between 1978 and 1983 cannot be tested. Data for the two periods are not directly comparable, as described above for walleye. There was a small (0.15 $\mu g\ g^{-1}$) but significant decline between 1986 and 1988, with the lower level also recorded in 1989. As in the other species, pike mercury levels remain substantially elevated 14 years after impoundment. In 1989, the mean pike mercury level in Rat Lake was a factor of 3 higher than the average for northern Manitoba lakes remote from hydro development (0.3-0.4 $\mu g\ g^{-1}$; Derksen and Green 1987) and was more than 3.5 times the pre-development level upstream in SIL (1971-1973, 0.26-0.32 $\mu g\ g^{-1}$; Bodaly et al. 1984).

4.2.2 Threepoint Lake

Threepoint Lake was flooded by the Churchill River diversion in 1976 (Figure 1). Lake level rose about 3.9 m and area increased 39% (G. McCullough, Department of Fisheries and Oceans, Winnipeg, MB, pers. comm.).

The post-impoundment peak in whitefish mercury occurred sometime between 1976 and 1981 (Figure 20). The timing and magnitude of the peak is unknown due to the lack of samples during this period. The level was constant, at about 0.2 $\mu g\ g^{-1}$, between 1983 and 1989. Although there are no pre-development whitefish mercury data for Threepoint Lake, the current level most probably

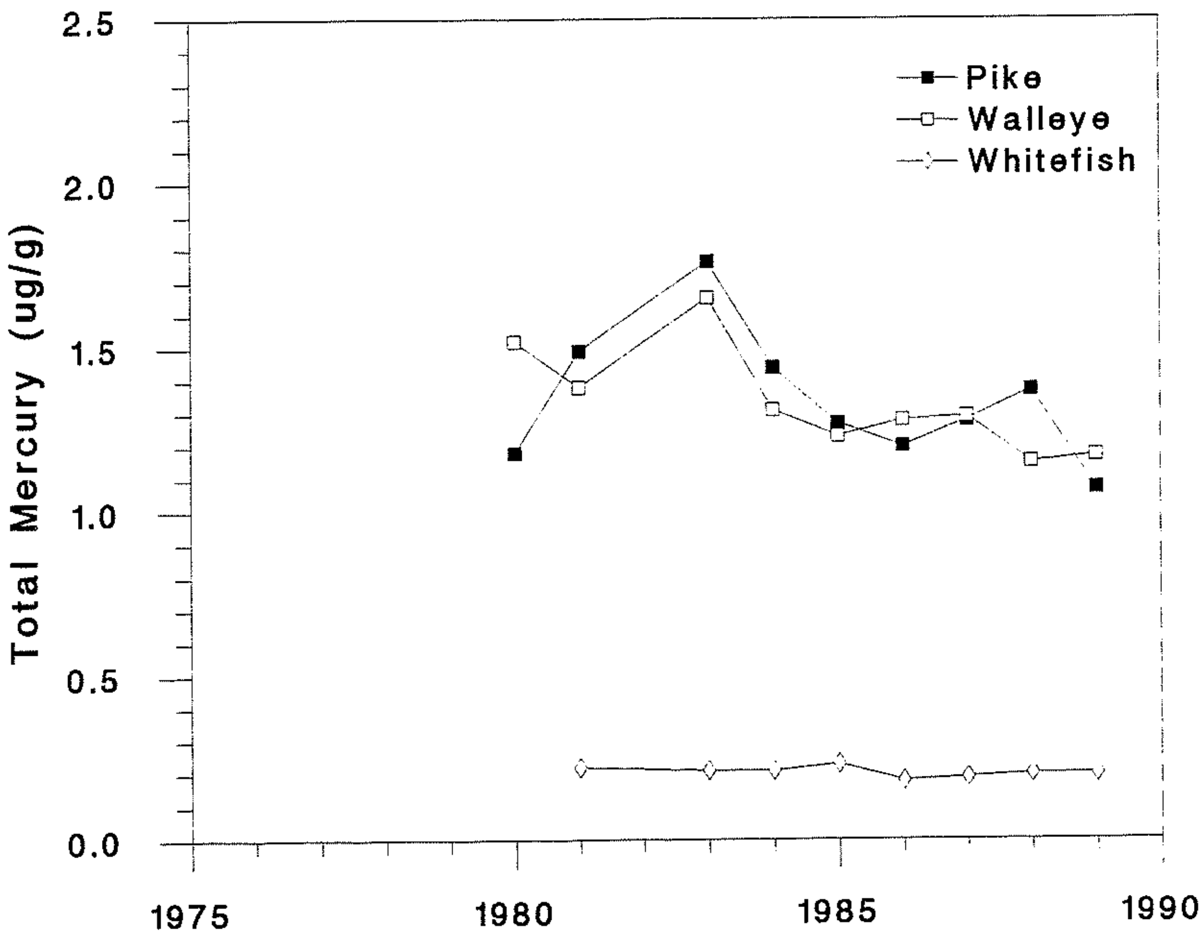

Figure 20. Mean total mercury concentrations in northern pike, walleye, and whitefish from Threepoint Lake, 1980-1989. Pre-1983 data from Green (1986). Later data are length-standardized means from Green (1990).

is elevated, as it is a factor of 2 higher than the average for northern Manitoba lakes remote from hydro development (<0.1 $\mu g\ g^{-1}$; Derksen and Green 1987) and 2.5 to 4 times the pre-development levels upstream in SIL (0.05-0.07 $\mu g\ g^{-1}$; Bodaly et al. 1984) and downstream in Wuskwatim Lake (0.08 $\mu g\ g^{-1}$ in 1970; Bodaly et al. 1984).

The post-impoundment peak in walleye mercury, 1.64 $\mu g\ g^{-1}$, appears to have occurred in 1983 (Figure 20). The mean level decreased significantly between 1983 and 1984, to 1.3 $\mu g\ g^{-1}$, and remained at this concentration through 1989. Twelve years after impoundment and Churchill River diversion, walleye mercury in Threepoint Lake was more than a factor of 3 higher than both the average for northern Manitoba lakes remote from hydro development (0.3-0.4 $\mu g\ g^{-1}$; Derksen and Green 1987) and the pre-development concentrations upstream in SIL (1971-1975, 0.19-0.30 $\mu g\ g^{-1}$; Bodaly et al. 1984) and downstream in Wuskwatim Lake (1970-1975, 0.25-0.40 $\mu g\ g^{-1}$; Bodaly et al. 1984).

The post-impoundment peak in pike mercury, 1.75 $\mu g\ g^{-1}$, also appears to have occurred in 1983 (Figure 20). The mean level declined significantly over the following 3 years, reaching 1.2 $\mu g\ g^{-1}$ in 1986. No further significant declines occurred between 1986 and 1989.

4.2.3 Stephens Lake

Stephens Lake was created in two stages in the autumns of 1970 and 1971 by the construction of the Kettle Generating Station on the Nelson River at Gillam, Manitoba (Water Resources Branch 1974). The depth of flooding at the dam is 30 m and total water area increased by about 300% (G. McCullough, Department of Fisheries and Oceans, Winnipeg, MB, pers. comm.)

The trend of slowly declining fish mercury levels after impoundment is evident in Stephens Lake pike and walleye but not in whitefish. There was no significant change in whitefish mercury between 1983 and 1989 (Figure 21). Averaging 0.16 $\mu g\ g^{-1}$ during this period, whitefish mercury in Stephens Lake was over 60% higher than the average for northern Manitoba lakes remote from hydro development (<0.1 $\mu g\ g^{-1}$; Derksen and Green 1987) and was double the mean level upstream in Split Lake during the same period (Ramsey 1991). Although there are no pre-development whitefish mercury data for Stephens Lake, the current levels clearly appear to be elevated, consistent with the demonstrably elevated levels in other species in the lake.

Walleye mercury in Stephens Lake declined significantly, from 1.37 $\mu g\ g^{-1}$ in 1983 to 0.81 $\mu g\ g^{-1}$ in 1984, and stabilized at this lower level from 1984 to 1989 (Figure 21). The timing and magnitude of the peak post-impoundment walleye mercury level will never be known with certainty because no samples were collected in the first 10 years after flooding. It is difficult to determine if the decline between 1983 and 1984 represented a real decrease or was an

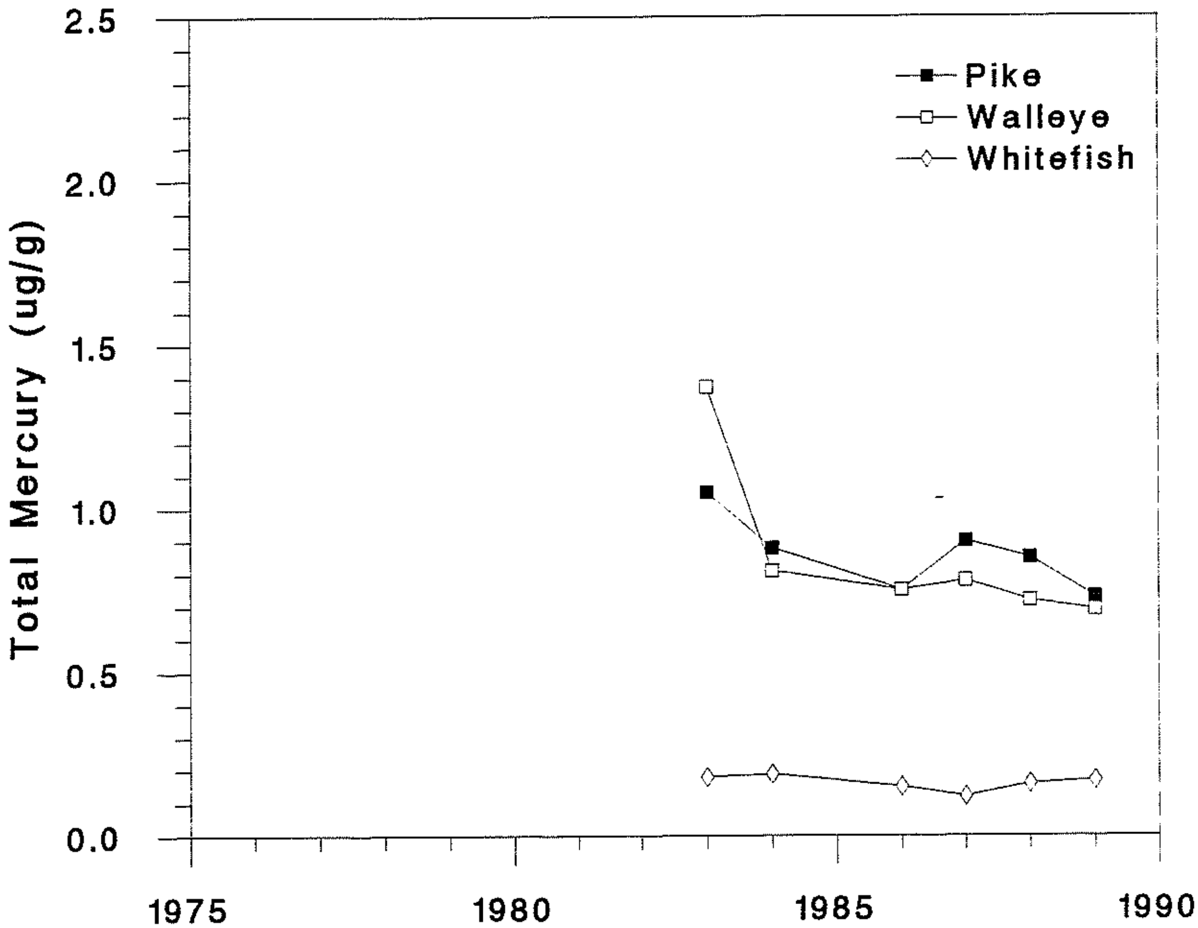

Figure 21. Mean total mercury concentrations in northern pike, walleye, and whitefish from Stephens Lake, 1983-1989. All data are length-standardized means from Green (1990).

artefact of a small (n=9), size-biased sample in 1983. As late as 1989, 18 years after impoundment, walleye mercury was 3 times higher than the pre-development level of 0.25 $\mu g\ g^{-1}$ reported for Moose Lake (n=1, Department of Fisheries and Oceans, Southern Operations Directorate, Winnipeg, MB), which was inundated in the formation of Stephens Lake.

Pike mercury also declined significantly, from 1.05 to 0.75 $\mu g\ g^{-1}$, between 1983 and 1986, and remained at this lower level through 1989 (Figure 21). The timing of the peak in pike mercury also will never be certain due to the lack of data for the 1971-1982 period. The standardized pike mercury level in 1989 was 4 times the pre-development concentration of 0.20 $\mu g\ g^{-1}$ reported for Moose Lake (n=4, Department of Fisheries and Oceans, Southern Operations Directorate, Winnipeg, MB).

4.2.4 Southern Indian Lake

The level of Southern Indian Lake was raised 3 m in August, 1976, to facilitate diversion of the Churchill River into the Rat-Burntwood and Nelson river systems at South Bay. Lake area increased approximately 9% due to impoundment (G. McCullough, Department of Fisheries and Oceans, Winnipeg, MB. pers. comm.)

Fish mercury levels have been monitored at a number of sites on SIL regularly between 1978 and 1988. This section will concentrate on the data for three species, northern pike, walleye, and whitefish, from four sites, Area 5, Area 4, the Channel, and Camp 9 (Figure 22). Data for other species or areas of the lake may be

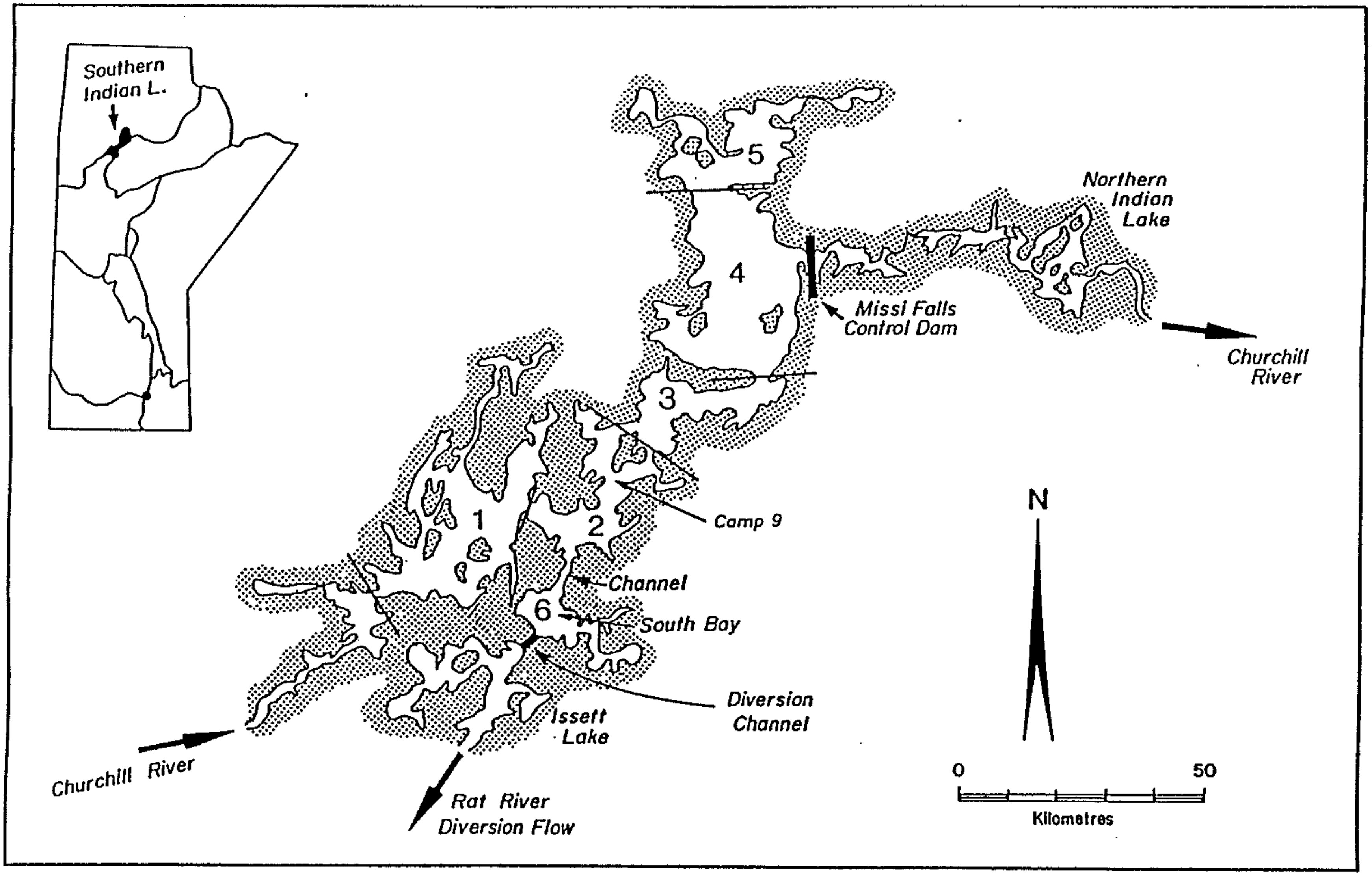

Figure 22. Location of Department of Fisheries and Oceans fish mercury sampling locations on Southern Indian Lake. From Strange et al. (1991).

found in Strange et al. (1991).

Mercury levels in all three species increased sharply after impoundment in 1976 (Figures 23, 24, and 25). Whitefish mercury peaked between 1978 and 1981, depending on location. Levels have since declined and, by 1988, had returned to near pre-impoundment concentrations in all surveyed regions with the exception of Area 5. Walleye mercury peaked between 1979 and 1981 (Figure 24). Levels have since declined from these peak values but, as late as 1988, mercury levels were still well above the pre-impoundment concentration. In contrast to the whitefish and walleye, the mean mercury concentration in northern pike has not yet started to decline at any site and may be continuing to increase (Figure 25).

4.2.5 Comparison of Temporal Trends in Fish Mercury

There are several differences in the temporal trends in fish mercury levels in SIL compared with sites on the CRD and Nelson River. The first difference is the clearly declining levels in SIL whitefish, in contrast to the other 3 lakes where no substantial or sustained decreases have occurred. Secondly, the decline in walleye mercury levels in SIL also has been larger in proportion to the initial increase than at the other sites, following the trend in whitefish mercury. Another difference is the occurrence of lower mercury levels in walleye than in pike from SIL, unlike the other sites. Finally, pike mercury in SIL has not declined at all, contrary to the trends in other species from SIL and to the slight downward trends in pike mercury at the other sites.

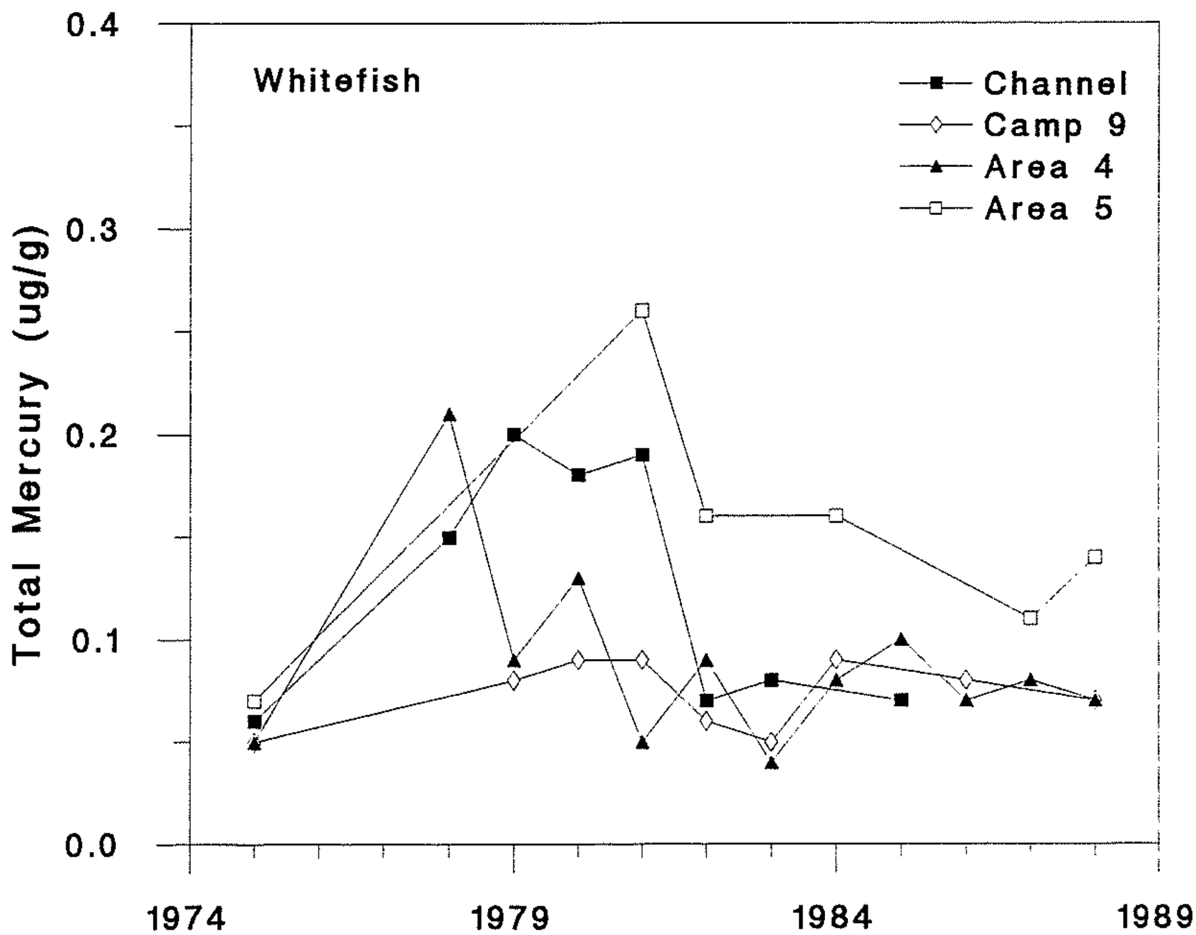

Figure 23. Mean length-standardized mercury concentrations in whitefish from sites on Southern Indian Lake, 1975-1988. All data from Strange et al. (1991).

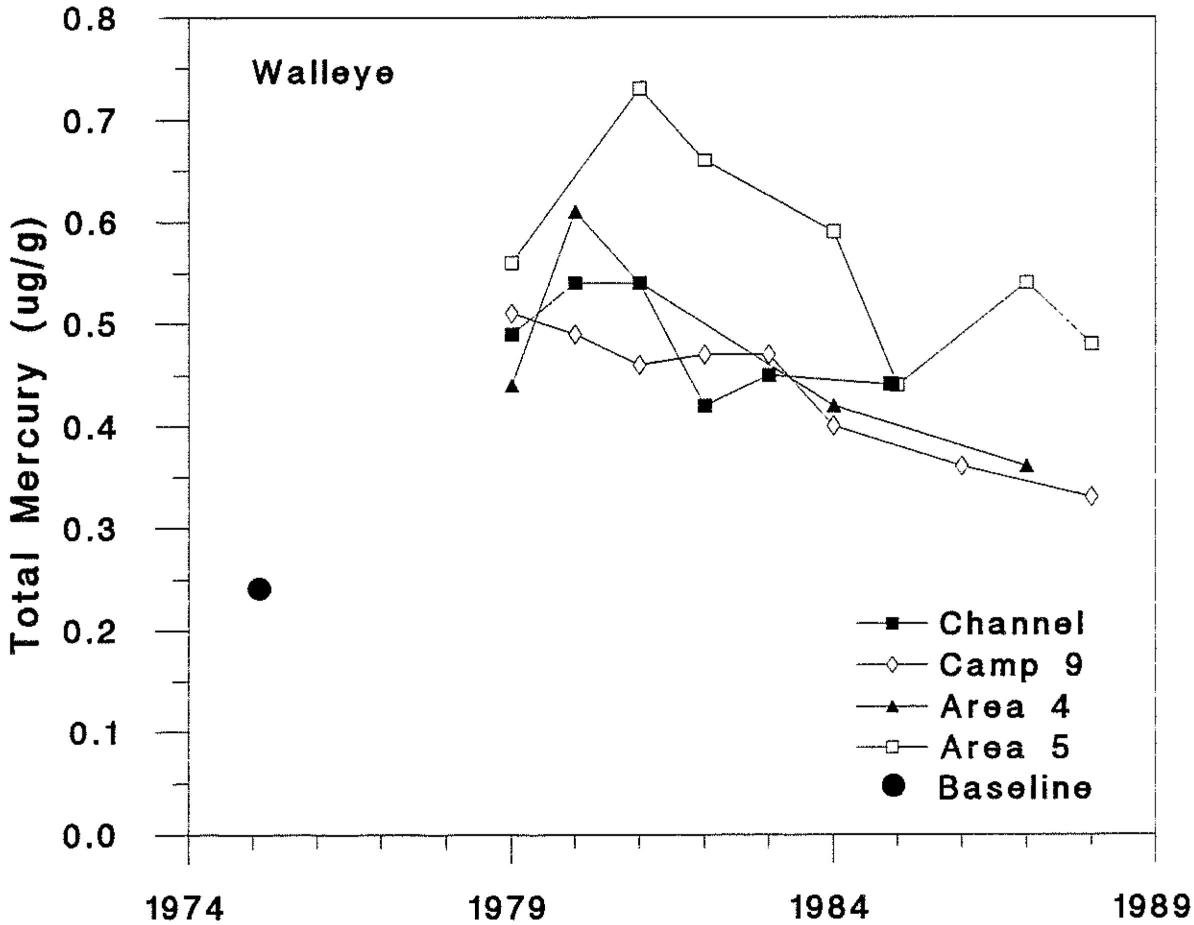

Figure 24. Mean length-standardized mercury concentrations in walleye from sites on Southern Indian Lake, 1975-1988. All data from Strange et al. (1991).

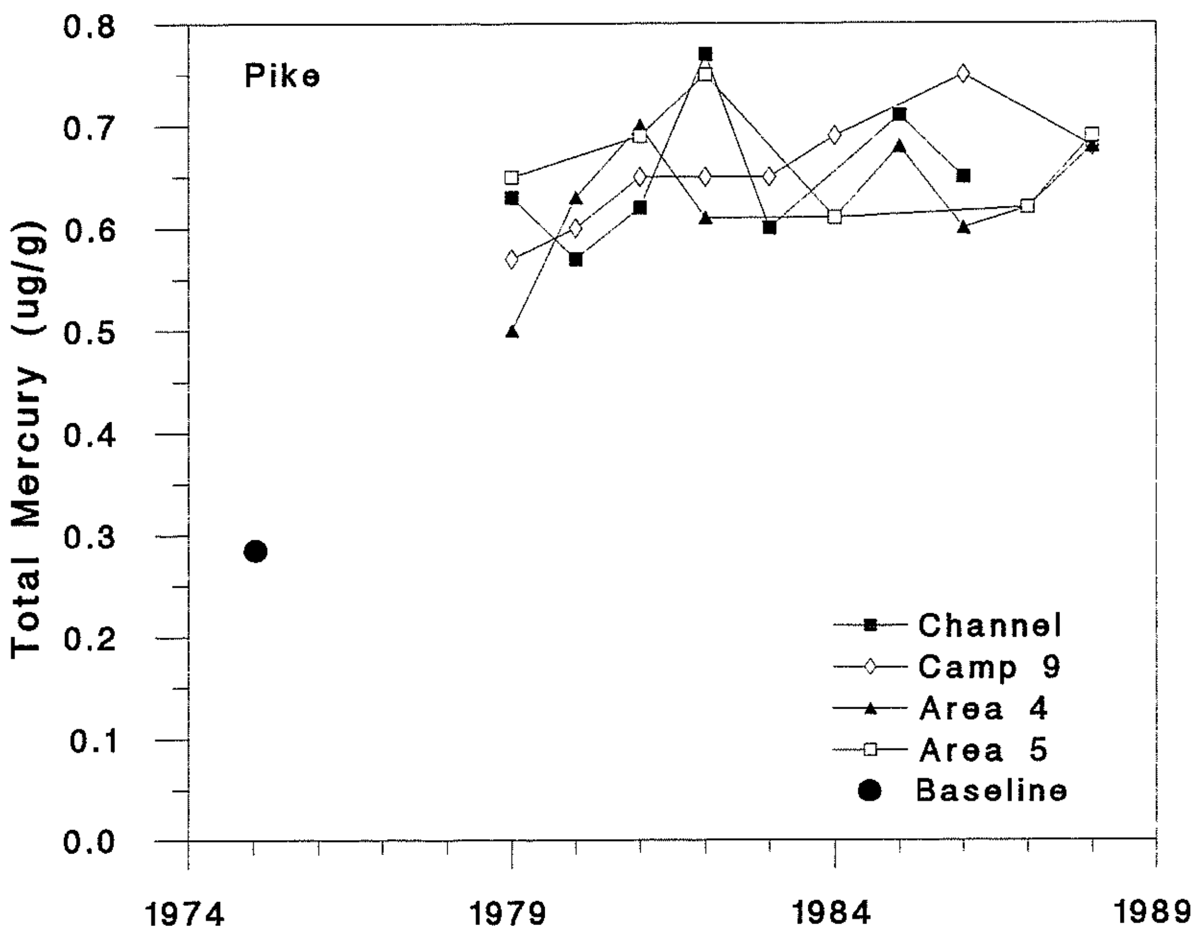

Figure 25. Mean length-standardized mercury concentrations in northern pike from sites on Southern Indian Lake, 1975-1988. All data from Strange et al. (1991).

These differences in temporal trends can be explained by two factors; (i) interaction of the species' different habitat preferences with the relatively small extent of flooding on SIL compared to other flooded lakes on the CRD and Nelson River and, (ii) the changes in limnological conditions which have occurred in Notigi Reservoir and Stephens Lake.

Pike prefer the nearshore zone (Inskip 1982), spending most of their time over the flooded terrain which is the site of elevated methyl mercury production. Net methylation in the flooded zone of SIL remains elevated, with no evidence of a decline since 1984 (see Section 4.1), and mercury levels in pike reflect this continued exposure. Given that the same types of materials were flooded in Notigi Reservoir, and these materials have been submerged for about the same length of time as in SIL, there is no reason to expect the ability of the flooded materials to stimulate methyl mercury production has decreased. Nevertheless, mercury levels in Rat Lake pike have declined. There must be some other explanation and a clue is provided by the timing of the decreased pike mercury levels in Rat Lake which coincided with the occurrence of elevated suspended sediment levels in Notigi Reservoir (McCullough 1990). Nearshore sedimentation is a consequence of increased shoreline erosion, as seen in SIL (Newbury and McCullough 1984; McCullough 1990). Sediment deposition reduces the area of exposed organic matter and, therefore, the area of reservoir sediments supporting elevated net microbial methylation, resulting in reduced methyl mercury availability and a lower mercury level in pike. The covering of organic matter with inorganic sediment appears to effectively prevent the stimulation of microbial methylation (Hecky et al. 1987a). Consistent with this hypothesis, a similar increase in turbidity, indicating an increase in shoreline erosion, has been noted in Stephens Lake (Berger and Ramsey 1991). The small declines in walleye mercury levels in Rat and Stephens lakes are consistent with the declines in pike mercury and are probably due to the same cause.

In contrast to pike, adult whitefish are most often found mid-lake in deeper, cooler water. The whitefish only move into the nearshore areas in the late autumn when water temperatures drop below 10 °C (Derksen and Green 1987). In SIL, where only about 10% of the lake is comprised of flooded terrain (G. McCullough, Department of Fisheries and Oceans, Winnipeg, MB, pers. comm.), this habitat preference places the whitefish over pre-existing lake sediments in the summer during the period of peak methyl mercury production. Net methylation in these pre-existing lake sediments has been at baseline levels since at least 1984 (Ramsey 1991). Methyl mercury produced in the flooded zone also appears to remain there, at least in Methyl Bay (Ramsey 1991). Thus, whitefish in much of SIL probably are not exposed to elevated methyl mercury levels during their daily activities.

This explanation accounts for the current low levels of mercury in SIL whitefish, but it does not account for the elevated levels which occurred immediately after impoundment. Either methyl mercury was exported offshore during this period or, more likely, the initial period of active shoreline erosion after impoundment resulted in organic matter being carried offshore where it could stimulate microbial methylation. Over time, eroding inorganic material would bury these offshore organics, reducing offshore methylation and, consequently, mercury levels in whitefish. In support of this hypothesis, Newbury and McCullough (1984) found organic matter of terrestrial origin as far as 300 m offshore (the maximum distance offshore that was sampled) and 250 m beyond the pre-existing lake shoreline. The area of exposed organic matter capable of stimulating microbial methylation clearly was much larger in the past than at present, and this explains the higher whitefish mercury levels in SIL immediately after impoundment. At the more extensively flooded sites such as Rat and Stephens lakes, where lake area occurs over flooded terrain, whitefish behaviour would not exclude them from exposure to flooded areas and the elevated methyl mercury levels found there. As a result, whitefish mercury levels in these lakes are higher than in SIL and decline more slowly.

Walleye are wide-ranging, moving into littoral areas at night and retreating to the deeper areas of the lake during the day (Derksen and Green 1987). This behaviour should reduce the exposure of walleye to the elevated methyl mercury levels in the flooded zone of SIL relative to the pike, accounting for the lower mercury levels in SIL walleye. In the more extensively flooded lakes, walleye behaviour does not provide an escape from flooded terrain, resulting in the similar mercury levels in pike and walleye from

Rat and Stephens lakes. The considerable decline in walleye mercury in SIL since 1980-81 can be accounted for in the same way that the decline in whitefish mercury is explained.

The fish mercury data for lakes on the CRD and Nelson River indicate declines are occurring at some but not all locations and are proceeding in a slow and discontinuous manner. Because the declines are slow and erratic in time and space, and the data record for all lakes is short relative to the rate of decline, it is difficult to base predictions of duration on the observed declines in fish mercury levels.

4.3 Methylation in Older Reservoirs

In addition to the annual measurements of microbial methylation and demethylation rates in SIL and Granville Lake, several older reservoirs were surveyed between 1986 and 1989. Reservoirs sampled in 1986 included the Ogoki Reservoir in northwestern Ontario, created in 1943, and Laurie Reservoir #2 situated on the Laurie River, flooded in the early 1950's (Figure 1). Sipiwesk Lake on the Nelson River, flooded in 1960 by the Kelsey Dam, was sampled in 1987. Sokatisewin Lake on the Churchill River in Saskatchewan, created in 1929 by the Island Falls Generating Station, was sampled in 1988.

A comparison of methylation and demethylation rates in the flooded zones of the SIL, Ogoki, and Laurie reservoirs in 1986 suggested that rates of both processes declined over time according to negative exponential models (Ramsey 1987a). These models were

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A comparison of methylation and demethylation rates in the flooded zones of the SIL, Ogoki, and Laurie reservoirs in 1986 suggested that rates of both processes declined over time according to negative exponential models (Ramsey 1987a). These models were originally formulated using an analytical approach. They have been reformulated as regression models in this report for comparison with later versions (Figures 26, 27, and 28). Curve shapes differ slightly from those originally presented, but the estimates of recovery time are essentially the same. Use of a regression approach permitted the mathematic description of the decline in methylation balance over time, with a second order polynomial providing the best fit.

Based only on the 1986 data, methylation rates were projected to return to the baseline level measured in Granville Lake 57 years after impoundment (Figure 26). Demethylation rates were estimated to return to baseline 44 years after impoundment (Figure 27) with the methylation balance returning to baseline after 53 years (Figure 28). These models were tested and modified over the following three years with additional data from Methyl Bay and the other older reservoirs.

Even with the additional 3 years of measurements, the models describing the evolution of methylation and demethylation rates and the methylation balance over time remain very similar to those based only on the 1986 survey data. The negative exponential equation has been retained for the methylation and demethylation models (Figures 29 and 30) because it consistently provided the best fit to the data. This model also is consistent with the cause of the reservoir mercury problem. The rate of decomposition of organic matter also follows a negative exponential model (Hodkinson 1975; Anderson et al. 1978). The estimated time required for the

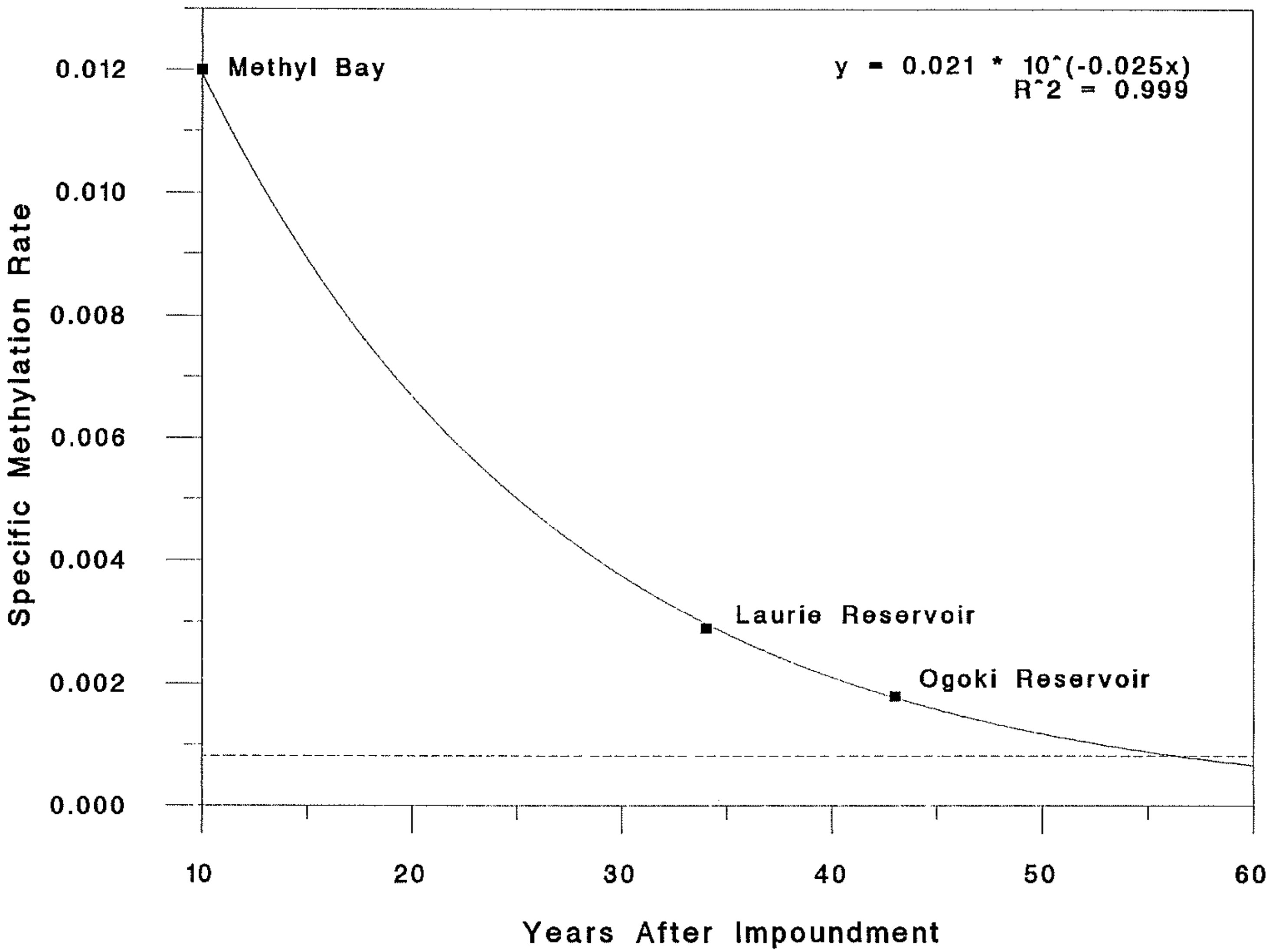

Figure 26. Relationship between the seasonal mean specific methylation rate (% added Hg methylated (g dry sediment)$^{-1}$ h^{-1}) in reservoir sediments and reservoir age for sites surveyed in 1986. Dashed line represents methylation rate in Granville Lake, a non-flooded reference lake. See Figures 1 and 5 for site locations. Data from Ramsey (1987a and b).

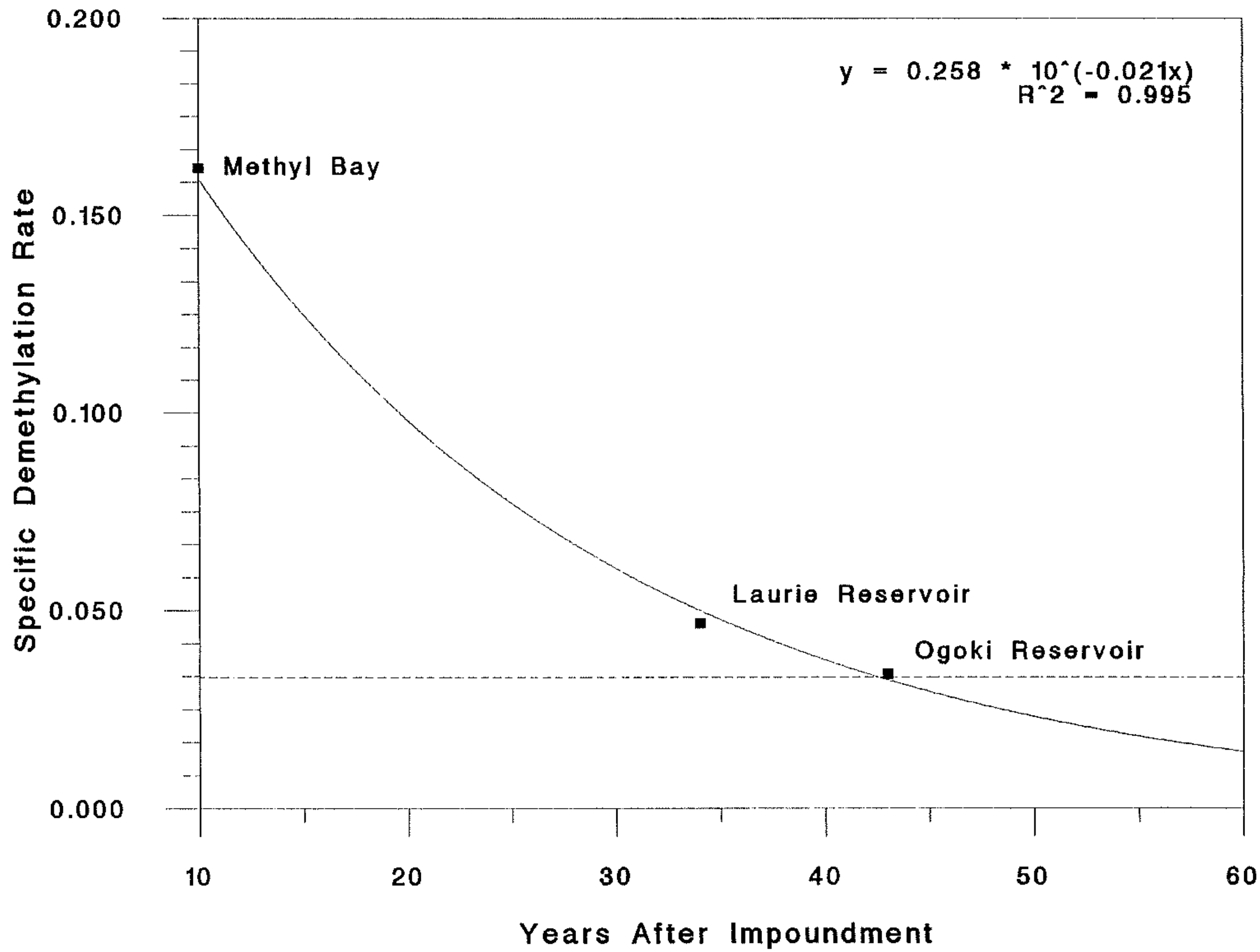

Figure 27. Relationship between the seasonal mean specific demethylation rate (% added Hg demethylated (g dry sediment)$^{-1}$ h^{-1}) in reservoir sediments and reservoir age for sites surveyed in 1986. Dashed line represents methylation rate in Granville Lake, a non-flooded reference lake. See Figures 1 and 5 for site locations. Data from Ramsey (1987a and b).

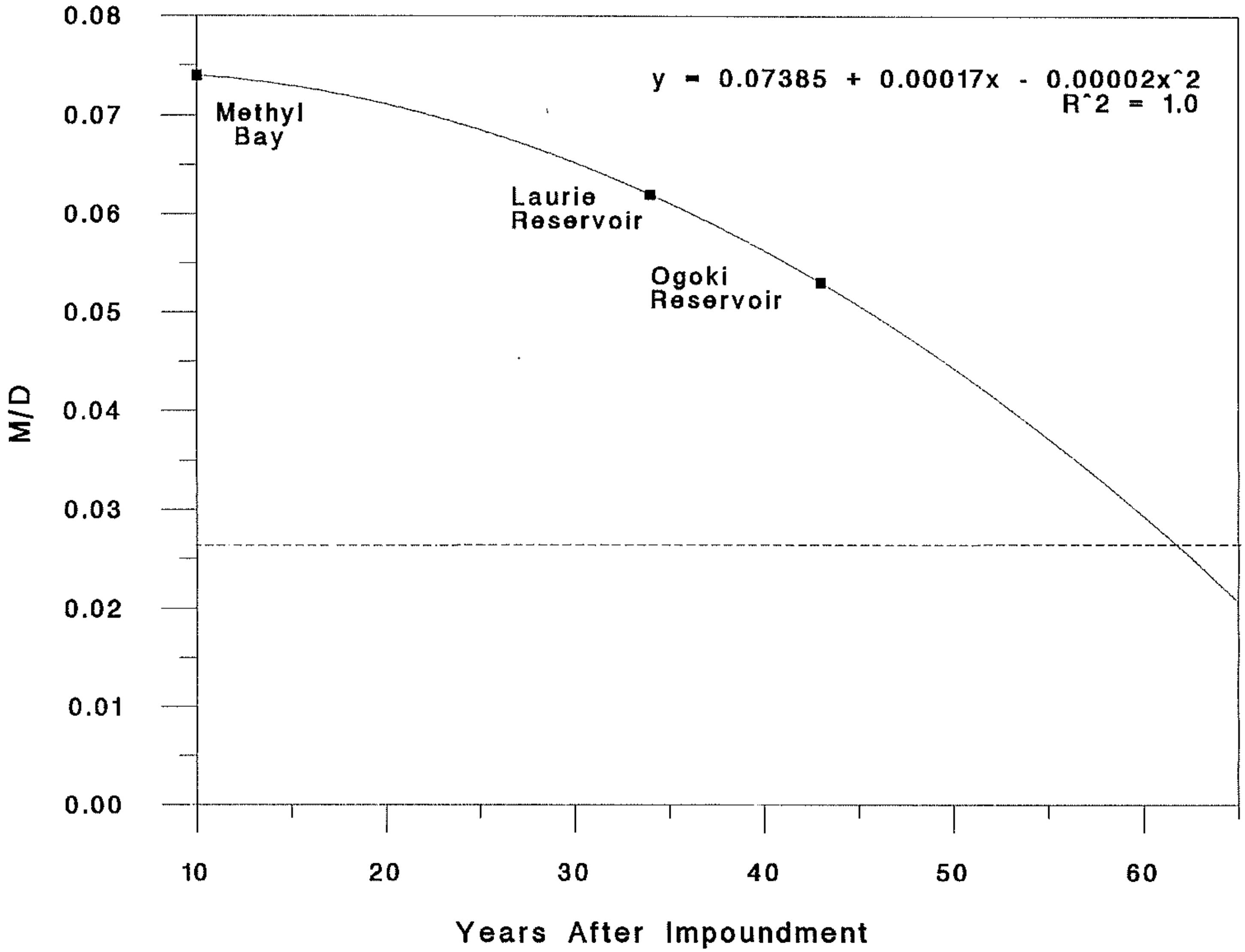

Figure 28. Relationship between the seasonal mean methylation balance (M/D) in reservoir sediments and reservoir age for sites surveyed in 1986. Dashed line represents methylation rate in Granville Lake, a non-flooded reference lake. See Figures 1 and 5 for site locations. Data from Ramsey (1987a and b).

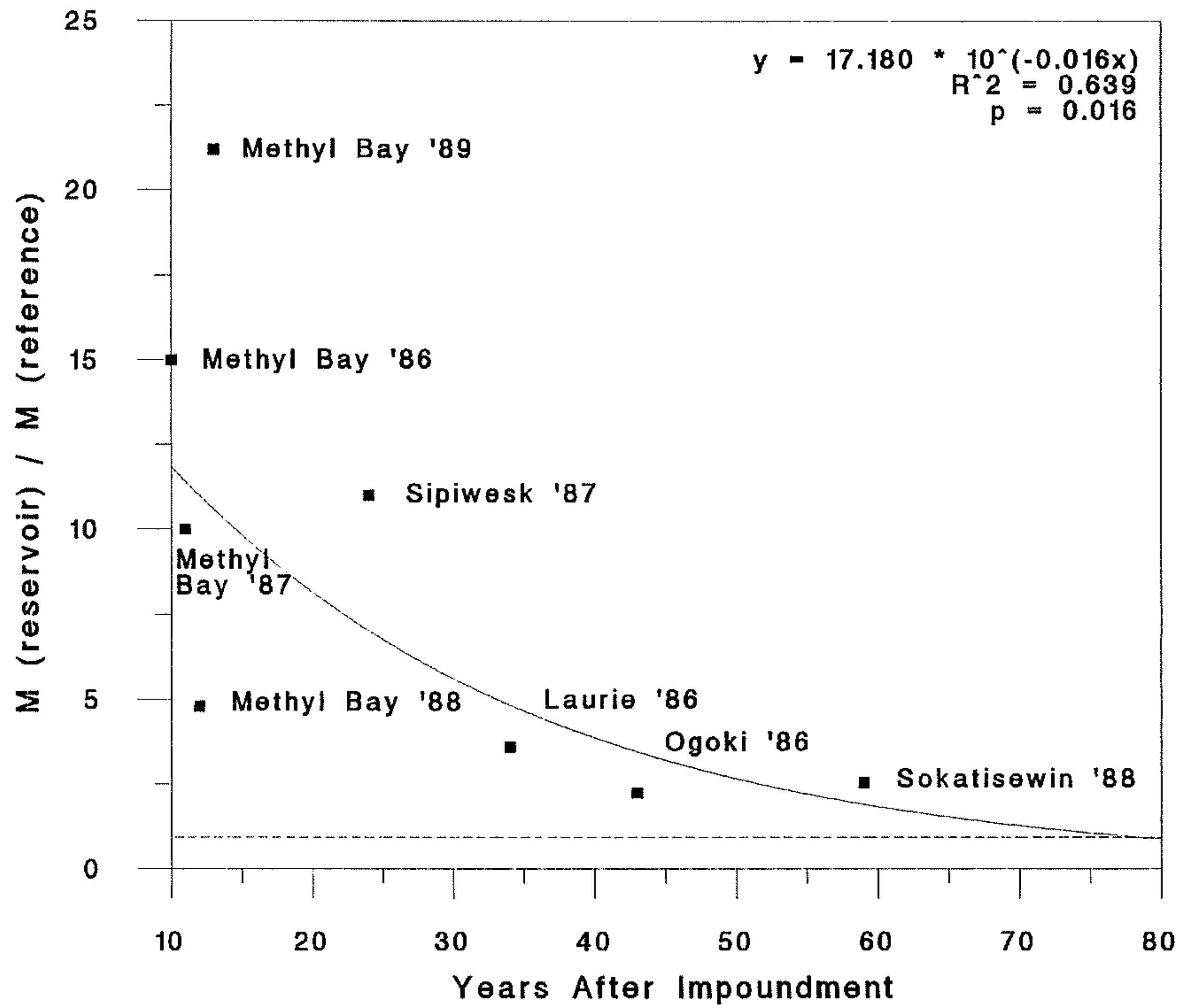

Figure 29. Relationship between the elevation in seasonal mean specific methylation rate (% added Hg methylated (g dry sediment)$^{-1}$ h^{-1}) in reservoir sediments and reservoir age for sites surveyed between 1986 and 1989. Dashed line represents equality of rates in reservoir and reference lake sediments. Granville Lake was the reference site in all years. See Figures 1 and 5 for site locations. Data from Ramsey (1987a and b, 1988a, 1989a, 1990).

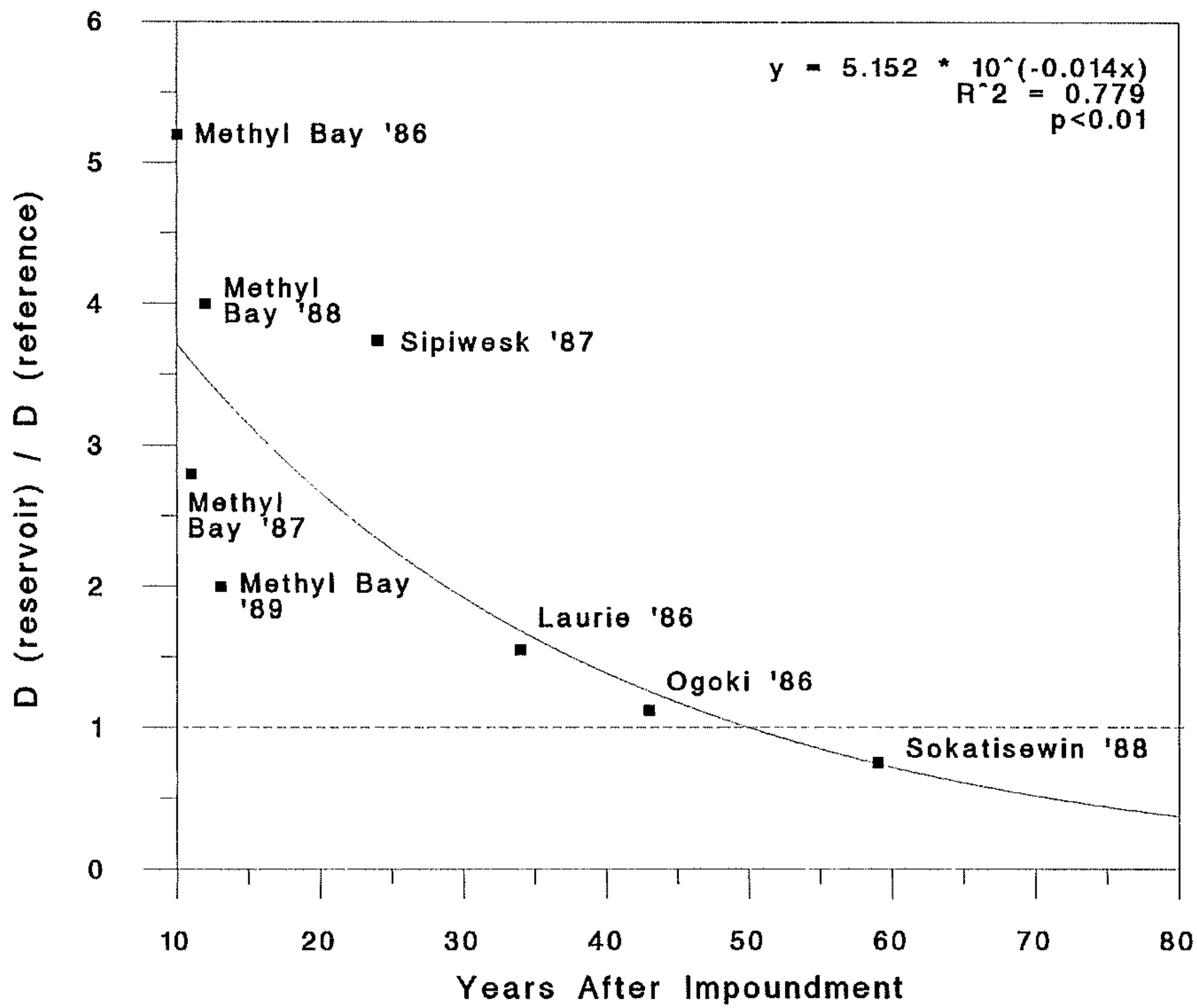

Figure 30. Relationship between the elevation in seasonal mean specific demethylation rate (% added Hg demethylated (g dry sediment)$^{-1}$ h^{-1}) in reservoir sediments and reservoir age for sites surveyed between 1986 and 1989. Dashed line represents equality of rates in reservoir and reference lake sediments. Granville Lake was the reference site in all years. See Figures 1 and 5 for site locations. Data from Ramsey (1987a and b, 1988a, 1989a, 1990).

return of methylation rates to the reference level is 78 years after impoundment, compared with 53 years for demethylation to return to baseline (Figures 29 and 30).

The methylation balance decline curve in Figures 31 and 32, was derived from the regression lines for methylation and demethylation in Figures 29 and 30. The least squares means of the elevated methylation and demethylation values were calculated at 10 year intervals for the period from 10 to 80 years after impoundment. These values were used to calculate the corresponding M/D values to which the curve in Figures 31 and 32 was fitted. As found for the 1986 data alone, a second order polynomial regression provided the best fit to the predicted M/D values. This model indicates there will be a slowly accelerating rate of decline in net methyl mercury production over time. The half-time (i.e., the period required for a 50% reduction) for the elevated methylation balance in the CRD is estimated to be 59 years, returning to the pre-development level 78 years after impoundment.

Although the basic structure of the models has remained the same over the course of the FEMP studies, there have been two important modifications. First, the dependent variables incorporated in the models have been modified from the measured specific methylation and demethylation rates (Ramsey 1987a and 1988a) to the elevation in methylation, demethylation, and M/D (Ramsey 1989a, 1990). Elevation is calculated as the seasonal mean value in flooded sediments from a given reservoir site divided by the seasonal mean value at the reference site. Granville Lake

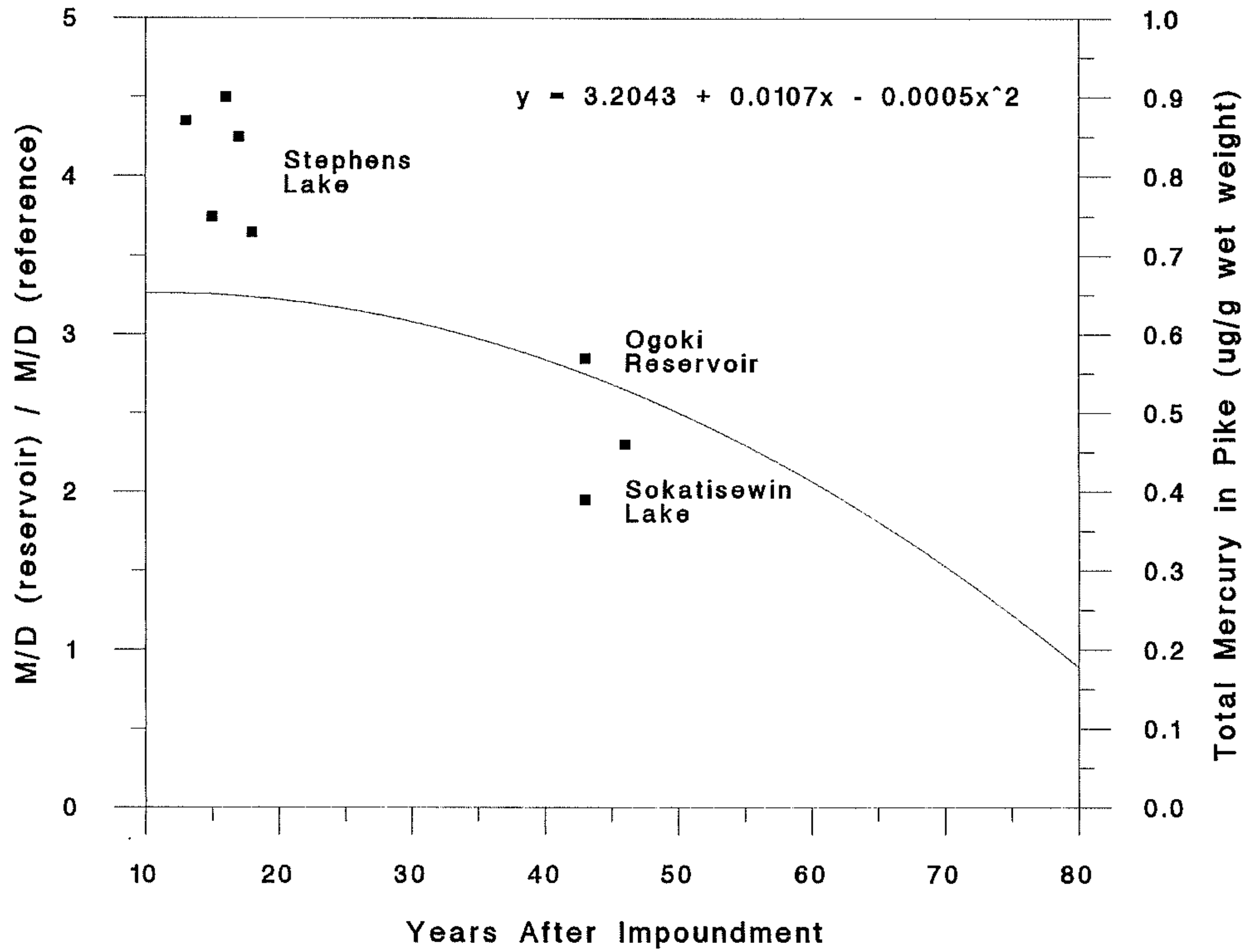

Figure 31. Relationship between elevation in methylation balance (M/D), mean mercury concentration in northern pike, and reservoir age for sites surveyed between 1986 and 1989. The M/D decay curve was calculated from the best-fit lines in Figures 29 and 30. Pike mercury data for Stephens Lake are length-standardized (550 mm) values from Green (1990). Ogoki Reservoir data are length-standardized (550 mm) values from G. Pope, Ontario Hydro, Environmental Studies and Assessments Dept., Toronto, Ontario. Sokatisewin Lake data are from commercial

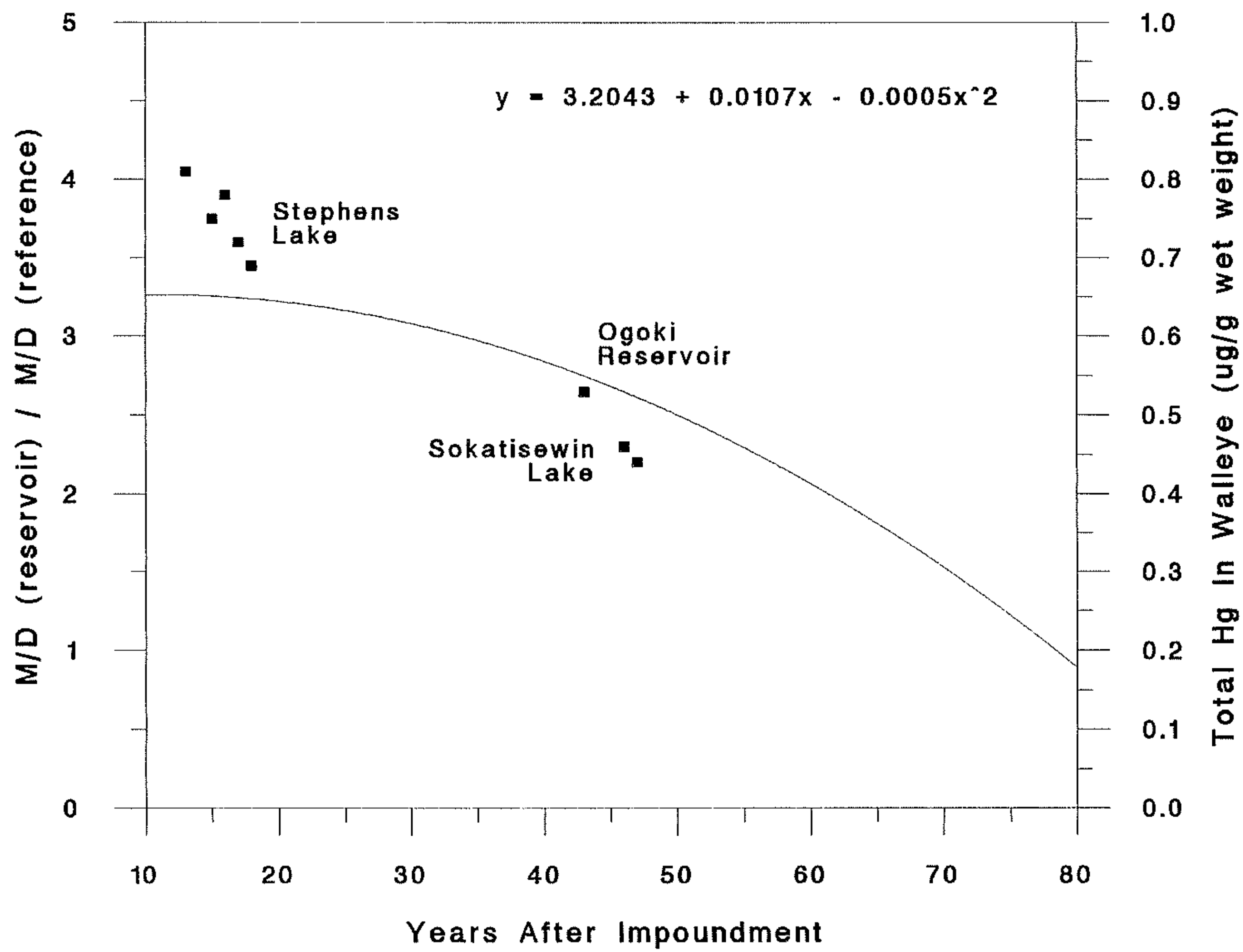

Figure 32. Relationship between elevation in methylation balance (M/D), mean mercury concentration in walleye, and reservoir age for sites surveyed between 1986 and 1989. The M/D decay curve was calculated from the best-fit lines in Figures 29 and 30. Walleye mercury data for Stephens Lake are length-standardized (400 mm) values from Green (1990). Ogoki Reservoir data are length-standardized (400 mm) values from G. Pope, Ontario Hydro, Environmental Studies and Assessments Dept., Toronto, Ontario. Sokatisewin Lake data are from commercial fishery

served as the reference site for all years. This standardization of the data was necessary to allow comparison of measurements made over several years because the specific methylation and demethylation rates are relative rather than absolute measures of methylating and demethylating activity.

The second and more recent change in the models is related to the nature of the mathematic expression employed to describe the declines in methylation and demethylation rates over time. The models presented in Ramsey (1989a and 1990) were based on regressions of ln-transformed elevation in methylation and demethylation against ln-transformed reservoir age (i.e., a ln-ln plot). The latest models presented in Figures 29 and 30 are the equivalent of regressions of ln-transformed methylation and demethylation against non-transformed reservoir age (i.e., a semi-ln plot). This second change was made because the revised models provided a slightly better fit to the methylation data. The change in the base exponent of the regression equations, from base-10 to base-e, had no effect on the shape or goodness-of-fit of the models.

All the seasonal mean values used in the models represent the time-weighted mean of five measurements, made at approximately 2 week intervals, for the period beginning June 24 and ending September 3. With the exception of the data for Methyl Bay in 1986, which represents a single station, all plotted values represent the average of data from three stations at each site.

The validity of the M/D decline model as a means of predicting the rate of decline of fish mercury levels was evaluated by overlaying pike and walleye mercury data on the predicted M/D decline curve. All fish mercury data were from reservoirs with similar characteristics which included; the same major fish species, the absence of an upstream reservoir within 130 km, similar extent of flooding, comparable regional or pre-development baseline mercury levels, and location within the boreal forest zone. These criteria were selected because they permit the assumption that peak post-impoundment fish mercury levels would have been similar. The distance of 130 km from the nearest upstream reservoir was based on the study of methyl mercury in water along the CRD, in which the export of methyl mercury from Notigi Reservoir was found to be restricted to a distance shorter than the 130 km between the Notigi Dam and Thompson (Section 2.4; Ramsey 1990).

The only Canadian reservoirs which appear to satisfy these requirements are Stephens Lake on the Nelson River (flooded 1970-71, 300% increase in lake area), the Ogoki Reservoir in north-western Ontario (flooded 1943, 266% increase in lake area, Ramsey 1987b), and Sokatisewin Lake on the Churchill River (flooded 1929, approximately 200% increase in lake area).

There is good agreement between the predicted elevation of M/D in a reservoir of given age, and the measured mercury concentration in fish (Figures 31 and 32). The slopes of the declines in M/D and fish mercury are similar and fish mercury levels appear to remain elevated for at least 47 years after impoundment. It has already been shown that mercury levels in Stephens Lake pike and walleye were 3-4 times above pre-development values 18 years after impoundment (Section 4.2.3). There also is strong evidence to suggest that levels in Sokatisewin Lake fish were elevated as late as 1976, 47 years after impoundment. No pre-development data exist for the lake, but data from other lakes on the Churchill River in northern Saskatchewan, upstream of Sokatisewin Lake, indicate fish mercury levels are considerably higher than is characteristic for the system (Table 8). Mercury levels in pike and walleye from upstream lakes ranged between 0.02 and 0.32 $\mu g\ g^{-1}$ and averaged 0.16 and 0.17 $\mu g\ g^{-1}$, respectively, for the period between 1970 and 1976. In the same period, mean walleye and pike mercury levels in Sokatisewin Lake ranged between 0.44 and 0.57 $\mu g\ g^{-1}$ and were 2.8 and 3 times higher than the respective upstream means. The lowest levels in Sokatisewin pike and walleye were, respectively, 60 and 38% above the highest levels measured upstream. Mercury levels in fish from Sokatisewin Lake have not been sampled since 1976 (G. McGregor, Department of Fisheries and Oceans, Inspection Services Branch, Winnipeg, MB, pers. comm.).

There is less certainty regarding the status of fish mercury levels in the Ogoki Reservoir. The river system has not been extensively sampled and there are no pre-development data for the lake. However, the mean methylation balance in Pikitigushi Lake, a nearby reference lake (Figure 5), was within 4% of that in Granville Lake in 1986 (Ramsey 1987b) and the mercury levels in

Table 8. Total mercury concentrations ($\mu g\ g^{-1}$ wet weight) in fish from lakes on the Churchill River in northern Saskatchewan. All data from commercial fishery samples as reported by Murray (1978). The contribution of a given lake-year to the overall mean was weighted by the number of samples in that year.

		Walleye		Northern Pike	
Lake	Year	Total Hg	n	Total Hg	n
Uskik	1971	0.21	5	0.18	2
	1972	0.08	1	0.29	1
Trade	1971	0.32	5	0.18	6
	1972	0.09	1	0.20	2
	1976	0.16	1	0.17	1
Pinehouse	1971	0.14	6	0.21	6
	1972	ND	ND	0.09	1
	1973	0.14	1	ND	ND
Ile-a-la-Crosse	1970	0.12	1	ND	ND
	1971	0.14	5	0.16	11
	1972	0.17	1	ND	ND
	1975	ND	ND	0.24	1
Churchill	1970	0.05	1	ND	ND
	1971	0.10	7	0.14	5
	1972	0.03	1	0.02	1
	1975	ND	ND	0.18	1
Frobisher	1970	0.08	1	ND	ND
	1971	0.18	5	ND	ND
	1975	0.12	1	ND	ND
Weighted Mean		0.16		0.17	
Sokatisewin	1972	ND	ND	0.57	6
	1975	0.46	1	0.46	1
	1976	0.44	2	ND	ND

ND No data.

walleye from the two lakes were the same (Pikitigushi, 0.27 $\mu g\ g^{-1}$; Ontario Ministry of Natural Resources, unpubl. data; Granville, 0.26 $\mu g\ g^{-1}$; Inspection Services, Department of Fisheries and Oceans, Winnipeg, MB.). This indicates the relationship between methylation balance and fish mercury in the Ogoki Reservoir area is the same as in the CRD, suggesting that M/D and fish mercury in the reservoir are elevated above natural baseline.

The estimates of recovery time presented above are considerably shorter than indicated previously, particularly for methylation and the methylation balance. Using the same data, I calculated a recovery time of approximately 135 years (Ramsey 1990), compared with the 78 year duration calculated above. This difference was due to the changes in the regression expression described above. The comparison of the decline in M/D with mercury levels in fish (Figures 31 and 32) suggests the revised model provides a better description of the time course.

4.4 Factors Affecting Duration

In the course of studies on mercury in the NFA area, a number of factors have emerged as having a significant modifying effect on the duration of mercury problems in northern reservoirs. These include the nature of the flooded organic matter with respect to its susceptibility to decomposition, the presence of a continuing source of fresh organic matter, and the erosion of shoreline clays.

The importance of the type of organic matter flooded is best illustrated by the time course of the elevated fish mercury levels in Cedar Lake following impoundment by the Grand Rapids Generating Station in 1965. Within 12 to 14 years after flooding, mercury concentrations in pike and walleye had declined to levels commonly found in northern Manitoba lakes remote from hydro development. The major difference between Cedar Lake and flooded lakes on the Rat-Burntwood and Nelson river systems was in the type of vegetation flooded. The area of Cedar Lake approximately doubled and about 80% of the flooded land was located in the area of the Saskatchewan River Delta (Green and Derksen 1980). Marsh macrophytes and sedges dominated the delta prior to flooding. These materials decompose more quickly than the boreal forest vegetation flooded in the other reservoirs, and contribute much (est. 90%) less organic matter per unit area of flooded land (Ramsey and Ramlal 1987b). These features of the Cedar Lake impoundment contributed to a much faster recovery than has occurred in any of the reservoirs in the NFA area.

Another factor to be considered is the way in which reservoir levels are regulated. In Sipiwesk Lake, levels are generally low in summer and high in winter. This is the reverse of the situation in most hydroelectric reservoirs, where water is stored during the summer and released in winter to meet increased power demands. Sipiwesk Lake was regulated in the usual manner until 1976, when the Jenpeg dam began operating, producing the reversed level regime. One of the results of this regime is exposure of the drawdown zone in summer when it can become recolonized by grasses and other fast-growing vegetation. These materials are flushed into the lake during periods of high water, becoming available to the methylating bacteria. A continuous source of fresh organic matter such as this should maintain methyl mercury production at a high level longer than would be expected in a reservoir with the usual level regime. This appears to be the case in Sipiwesk Lake, where the specific methylation rate in the flooded zone was at the high end of the expected range in 1987 (Ramsey 1989a).

The long-term impact of eroding glacial clays on the duration of the mercury problem in the CRD remains to be established. It was suggested in Section 4.2.5 that the decreases in fish mercury levels which have occurred in Rat and Stephens lakes coincided with increases in shoreline erosion. It was speculated the eroding clays were covering some of the flooded organic matter, effectively reducing the area of reservoir sediments supporting elevated methylation and resulting in lower mercury levels in fish. There is insufficient information to determine if this process will cause a continuing reduction in fish mercury levels, accelerating recovery of the mercury problem.

4.5 Duration of Mercury Problems in other Regions

The projected duration of the mercury problem in SIL and along the CRD, 78 years following flooding, is much longer than the predicted duration for the problem in the Complexe La Grande in northern Quebec. Verdon (1990) estimated mercury levels in northern pike from the Complexe La Grande will return to the baseline level 20-30 years after impoundment. This estimate is based on a comparison of mercury concentrations in pike from reservoirs in

Quebec and Labrador ranging in age from 3 to 67 years (Table 9). There is clear evidence of elevated levels continuing in reservoirs up to 16 years of age, the oldest reservoir in the data set for which there are pre-development data. The mercury concentration in 700 mm pike from Smallwood Reservoir in Labrador, at 1.11 $\mu g\ g^{-1}$ in 1988, was still 2.6 times higher than the pre-development level after 16 years of impoundment and had declined just 0.16 $\mu g\ g^{-1}$ in the preceding 10 years.

The estimated recovery time for the mercury problem in the Complexe La Grande relies heavily on data from older reservoirs for which no pre-development data exist. Mercury levels in pike from reference lakes adjacent to the older reservoirs were instead used as an indication of pre-development values. Only 1 reference lake was sampled for each of the older reservoirs, and no rationale for the selection of these reference lakes has been presented. Curiously, mercury levels in fish from the reference lakes were extraordinarily high, ranging between 1.06 and 1.38 $\mu g\ g^{-1}$ for 700 mm northern pike, compared with 0.43 to 0.63 $\mu g\ g^{-1}$ in the reservoirs for which pre-development data were available (Table 9). Without knowledge of the rationale used in selection of the reference lakes, and considering the extremely high mercury levels reported for the lakes, it is difficult to accept them as valid indicators of pre-development fish mercury levels. There is considerable natural variation in fish mercury levels among lakes such that, in the absence of pre-development data for a given site, very different conclusions can be drawn depending on the reference

Table 9. Total mercury concentrations ($\mu g\ g^{-1}$) in northern pike (700 mm) in reservoirs of varying ages in Quebec and Labrador. Natural Hg levels for LG-2 and Smallwood reservoirs are pre-development values. For other sites, natural levels are concentrations in a single nearby reference lake. Augmentation factor calculated as Total Hg/Natural Hg. Data from Verdon (1990).

Reservoir	Age (Years)	Total Hg ($\mu g\ g^{-1}$)	Natural Hg Levels ($\mu g\ g^{-1}$)	Augmentation Factor
LG-2 (1982)	3	1.32	0.63	2.1
LG-2 (1984)	5	2.66	0.63	4.2
LG-2 (1986)	7	2.30	0.63	3.7
LG-2 (1988)	9	3.14	0.63	5.0
Smallwood	6	1.27	0.43	3.0
Smallwood	16	1.11	0.43	2.6
Manic-5 (1985)[1]	20	1.94	1.23	1.6
Dozois	37	1.43	1.38	1.0
Decelles	44	0.99	1.38	0.7
Baskatong	59	1.37	1.06	1.3
Des Quinze	62	1.13	1.38	0.8
Gouin	67	1.19	1.22	1.0

1 Flooding of Manic-5 occurred from 1964 to 1970.

lake which is selected. For example, in 11 non-flooded reference lakes within 60 km of SIL, mercury levels in walleye range from 0.23 $\mu g\ g^{-1}$ in Gauer Lake, which is similar to the pre-impoundment value for SIL, to 0.86 $\mu g\ g^{-1}$ in Melvin Lake (Rannie and Punter 1987), which is higher than the peak post-impoundment walleye mercury value for SIL (Figure 24). A similar degree of inter-lake variation should be expected in Quebec as well, making any conclusions based on a single reference lake rather tenuous. Thus, it appears there is little justification for the claim that fish mercury levels in the Complexe La Grande will return to pre-development values within 20-30 years of impoundment.

The lack of pre-development fish mercury data for reservoirs in excess of 20 years of age makes it necessary to infer pre-development concentrations from nearby reference lakes. The substantial natural variation in fish mercury levels among lakes makes it difficult to arbitrarily select a single lake as representative of baseline levels. Considerably more data on baseline mercury levels in lakes around the older (>20 years of age) Quebec reservoirs are required before it can be concluded the reservoirs have recovered.

There is little reason to expect that the fish mercury problem in the Complexe La Grande will be of substantially shorter duration than the problem on the CRD. The two developments are situated in very similar biophysical settings and the same factors appear to be regulating the microbial methylation balance (Ramsey 1989b). The only major biophysical difference is the general absence of glacial clays in the La Grande area, resulting in little continuing shoreline erosion. This difference is not expected to shorten the duration in La Grande. No active shoreline erosion is occurring in Methyl Bay on SIL and neither of the two oldest reservoirs sampled in the FEMP studies, Ogoki and Sokatisewin Lake, is situated in an area of glacial clay deposits. Nevertheless, fish mercury levels in the older reservoirs fit the M/D decline curve (Figures 31 and 32).

The predicted duration of the mercury problem in the CRD also is considerably longer than reported for reservoirs in Finland. Verta et al. (1986) found a half-life of 15-20 years for elevated pike mercury levels in 20 reservoirs, ranging from 3 to 20 years in age. In most of the reservoirs, pike mercury was near the regional average of 0.56 $\mu g\ g^{-1}$ within about 15 years of impoundment. However, levels in some reservoirs remained high (>1.0 $\mu g\ g^{-1}$) 16-18 years after impoundment. The major difference between the Finnish reservoirs and those in northern Manitoba, and the likely explanation for the shorter time-course in Finland, is in the level management regimes. The Finnish reservoirs are typically shallow, with mean depths between 1.4 and 7.0 m, and are heavily regulated. The ratio of drawdown to mean depth ranged between 0.4 and 2.29 and averaged 1.16. Most are run off in the winter until almost empty and are refilled by the spring floods. The flooded materials are therefore subjected to substantial physical disturbance by ice and wind which would accelerate the processing and removal of organic matter from the reservoir. This physical processing does not occur to the same extent in northern Manitoba reservoirs due to the much more limited range of level fluctuation relative to mean depth. The maximum annual range of levels on SIL is approximately 1 m compared to a mean depth of 9.8 m (Newbury et al. 1984) for a drawdown ratio of 0.1.

5.0 Toxicity of Elevated Mercury Levels to Biota

Some concern has been expressed by users of fish and aquatic furbearer resources along the CRD about the possible effects of the elevated mercury levels on animal physiology and behaviour (L. Linklater, Nelson House, MB, pers. comm.). The available information indicates it is unlikely that physiological effects on fish, mink, or otter are now occurring or have occurred in the past, even at the highest concentrations measured on the CRD. Mercury concentrations in other mammals and birds from the NFA area are unknown, so it is not possible to assess the potential for physiological effects in these groups.

Laboratory toxicity studies on brook trout (*Salvelinus fontinalis*) indicate that whole-body mercury concentrations would have to exceed 3 to 5 $\mu g\ g^{-1}$ before physiological effects should be expected. McKim et al. (1976) found levels up to 2.7 $\mu g\ g^{-1}$ had no effect on growth or survival of juvenile or adult brook trout nor on reproduction, while levels in the range of 5 to 7 $\mu g\ g^{-1}$ eventually produced symptoms of toxicity and death. Olson et al. (1975) found concentrations up to 10.9 $\mu g\ g^{-1}$ in fathead minnows (*Pimephales promelas* Rafinesque) had no effect on the survival, growth, behaviour, or general appearance of the fish. Lockhart et al. (1972) found differences in the biochemical profiles of northern pike (*Esox lucius*) from Clay Lake, Ontario, with mercury levels in the range of 6.3 to 16.0 $\mu g\ g^{-1}$, when compared with pike from Heming Lake, Manitoba, with mercury levels of about 0.3 $\mu g\ g^{-1}$. These differences indicated the Clay Lake fish were stressed, but this could not be attributed directly to the high mercury levels because the lakes differed in other ecological factors.

Mercury concentrations in fish along the CRD presently are below the levels which have been shown to produce physiological effects. Mean mercury concentrations in pike and walleye from Rat and Threepoint Lake are in the range of 1.0 to 1.2 $\mu g\ g^{-1}$ and the maximum level found in an individual fish in 1989 was 3.31 $\mu g\ g^{-1}$ in a pike from Rat Lake (Green 1990). In the past, mercury levels in some of the largest fish may have approached the upper limit of the no effects range. In 1983, the highest mercury level measured in an individual walleye from Rat Lake was 4.1 $\mu g\ g^{-1}$ (Green 1990). The highest mercury concentration ever found in an individual fish from the CRD was 4.73 $\mu g\ g^{-1}$ in a pike from Karsakuwigamak Lake captured in 1984 (Green 1986). Considering that laboratory studies required higher fish mercury concentrations than the highest recorded in any single fish from the CRD to produce any measurable physiological effects, and that no clear evidence of physiological effects has been found in the field at even higher concentrations than in laboratory studies, there appears to be no justification for the examination of mercury-related physiological effects in the CRD in any more detail.

Kucera (1987) reported that mean mercury concentrations in tissues of mink and otter also were well within the non-toxic ranges for these species. The highest concentrations measured in individual animals approached the upper limit of the non-toxic range, as in fish.

6.0 Mitigation Measures

The following actions have been identified in the literature as possible approaches to the mitigation of aquatic mercury problems or rehabilitation of closed fisheries:

- selenium additions
- clay resuspension (increased turbidity)
- increase fish growth
    - intensive fishing
    - nutrient additions
- lake liming
- inhibition of methylation/stimulation of demethylation
- make use of species or size-classes of fish with lower mercury concentrations

6.1 Selenium Additions

The addition of selenium has been suggested as a potentially successful means of reducing fish mercury levels both in Canada and Scandinavia. Selenium has been shown to reduce the toxicity of several heavy metals, including mercury (Frost and Lish 1975, Vokal-Borek 1979), and can reduce the severity of inorganic and methyl mercury poisoning. Turner and Rudd (1983) found selenium additions also can reduce mercury bioaccumulation in aquatic food webs. These decreases can be substantial. Paulsson and Lundbergh (1990) reported a 50% decrease in pike mercury levels, and a 90% decrease in perch and roach mercury after less than a year of treatment with selenium at 3-5 $\mu g\ L^{-1}$.

A serious difficulty with the use of selenium is that the levels in water which are required to produce a significant reduction of mercury levels in fish (1-5 $\mu g\ L^{-1}$: Turner and Rudd 1983, Paulsson and Lundbergh 1990) come uncomfortably close to the Canadian limit for drinking water (10 $\mu g\ L^{-1}$; Health and Welfare Canada 1989). It is unlikely that selenium additions will be a viable means of ameliorating the mercury problem in the CRD for the foreseeable future unless the effective level can be substantially reduced or it can be conclusively demonstrated that the levels which have been reported to be effective are non-toxic to humans or aquatic life.

6.2 Increased Turbidity

Sediment resuspension is the only remedial measure which has been tested for application on the CRD. This method has been suggested as a possible mitigative action for the mercury problem in the English-Wabigoon river system in northwestern Ontario on the basis of a limnocorral experiment which indicated the method could effectively reduce mercury levels in fish (Rudd and Turner 1983). A similar study in SIL found otherwise. Increased suspended sediment concentrations did not significantly reduce mercury concentrations in fish and caused slight increases in some cases (Hecky et al. 1987a).

The conflicting results of these studies could be a reflection of the different causes of the problems in the two systems; increased inorganic mercury loading in the English-Wabigoon compared with increased organic matter loading in the CRD. The suspended sediments were thought to reduce mercury accumulation in fish by binding inorganic mercury, making it unavailable for methylation (Rudd and Turner 1983), and possibly binding methyl mercury as well, making it unavailable for uptake by fish. With the problem in the CRD apparently not due to increased inorganic mercury availability, control of this variable should not be expected to have a significant effect on methylation.

The discrepancy may also be a result of different experimental approaches. In the English-Wabigoon study, ^{203}Hg was added as a radiotracer and only ^{203}Hg was measured in the fish. An increase or decrease in ^{203}Hg concentration in tissue was assumed to be indicative of a change in total mercury bioaccumulation. The radiotracer also was added in the SIL experiments, but bioaccumulation rates of both total mercury and radio-labelled mercury were measured. The SIL study indicated that, in short term experiments, radiomercury uptake or loss was not necessarily indicative of total mercury uptake or loss (Hecky et al. 1987a). Thus there is serious question regarding the efficacy of sediment resuspension as a mitigating measure in any system, and it certainly does not seem to be applicable in the CRD. Indeed, the mercury problem developed at most locations in spite of increased suspended sediment concentrations. Although increased turbidity was not effective in reducing mercury bioaccumulation by fish, the covering of organic matter with inorganic sediments should still be effective in reducing microbial methylation (Hecky et al. 1987b).

6.3 Increase Fish Growth

The application of measures to increase fish growth as a means of reducing the mercury concentration in tissues has been suggested as a way of mitigating mercury problems by a number of authors (e.g., Beijer and Jernelov 1979; Jones et al. 1985). It is thought that if fish growth can be increased without causing a proportionate increase in methyl mercury availability, then the mercury concentration in fish flesh will be reduced. If mercury burden is related to methyl mercury availability, then the burden will be diluted in faster growing fish leading to lower mercury concentrations. This biodilution concept has been criticized, but the majority of evidence in the literature suggests the concept has merit at least under some circumstances. Two methods of increasing fish growth, their impact on mercury levels in fish and potential utility in the CRD are discussed below.

6.3.1 Intensive Fishing

Intensive fishing has been suggested as a means of reducing fish mercury levels by increasing fish growth rates, with some positive effects demonstrated in Finland. Iivonen and Verta (1990) intensively fished Hakojärvi, a small (0.17 km^2) forest lake in southern Finland in 1984 and 1985. More than half the fish stock, 29.5 kg ha^{-1}, mainly roach (*Rutilus rutilus* L.), was removed from the lake. The mean standardized (1 kg) mercury concentration in northern pike (*Esox lucius*) declined from 0.97 $\mu g\ g^{-1}$ before harvesting to 0.45 $\mu g\ g^{-1}$ in 1987. Similarly, the mean standardized (0.5 kg) level in burbot (*Lota lota*) decreased from 0.56 to 0.19 $\mu g\ g^{-1}$ and the mean level in roach (50 g) dropped from 0.36 to 0.25 $\mu g\ g^{-1}$ during the same period. The mercury concentration in perch (*Perca fluviatilis*) did not change between harvest and 1987. Since 1987, mercury levels in pike, burbot, and perch have increased, although mercury in pike and burbot had not returned to pre-harvest levels by 1989. Growth rates increased in all species, with the largest increase in pike and the smallest in burbot. Growth rates of perch and roach increased initially but returned to pre-harvest rates within three years. The reduced fish mercury concentrations were attributed to the increased growth rates (pike, roach) and altered diet (burbot).

Recent research has shown that a large amount of the methyl mercury in a lake, about 30%, is tied up in the fish (J. Wiener, Wisconsin Department of Natural Resources, La Crosse, Wisconsin, pers. comm.). This suggests that the removal of a significant portion of the fish community, through intensive fishing, might also reduce the size of the methyl mercury pool, further contributing to the reduction of mercury levels in the remaining fish.

Continued research on the mechanism by which intensive fishing reduces mercury levels appears to be warranted.

Although intensive fishing would appear to be an effective means of reducing mercury concentrations in fish, at least for periods of 3 to 5 years, its usefulness as a mitigating measure in northern Manitoba reservoirs remains to be established. An important limitation is the impact of the harvest on fish populations and their ability to support further harvest afterward.

Intensive fishing also is subject to some practical limitations. The lakes must be small enough for it to be feasible to substantially reduce fish standing stocks, and the potential for immigration of contaminated fish from other parts of the system must be limited. Any trophy sport-fishery on the lake also would be lost. These restrictions would exclude most of the lakes comprising Notigi Reservoir, with the possible exception of Central Mynarski Lake. Lakes below the Notigi Dam are better candidates for manipulation, provided fish from Notigi Reservoir do not contribute significantly to downstream stocks. This remains to be established.

Another potential use of intensive fishing is in rehabilitation of the "inland" lakes with naturally high mercury levels which are common in the NFA area. These lakes are generally smaller and better isolated than those on the CRD, making them more suitable for manipulation. Impacts on sport fisheries also are more easily avoided. Because of these characteristics, the inland high mercury lakes might be a better site to test the effectiveness of intensive fishing than lakes on the CRD.

6.3.2 Nutrient Additions

Another means of increasing fish growth, and potentially lowering muscle mercury concentrations, is the stimulation of algal productivity through inorganic nutrient additions. The best evidence that this approach might work is reported by D'Itri at al. (1971). They compared mercury concentrations in stocked rainbow trout from an **oligotrophic** and a **eutrophic** lake in northern Michigan. The mercury concentration in fish from the eutrophic lake was less than half the level in fish from the oligotrophic lake. Fish of a given age from the eutrophic lake weighed twice as much and were 27% longer than fish from the oligotrophic lake, suggesting the higher growth rate was responsible.

An experimental investigation of the effect of nutrient additions on fish mercury levels in SIL was inconclusive. Hecky et al. (1987a) increased **primary productivity** in a limnocorral by a factor of 2, with no change in mercury bioaccumulation by fish compared to a reference enclosure. Fish growth was not affected by the increased productivity so this experiment neither confirms nor refutes the hypothesis of growth dilution.

Regardless of its potential impact on fish mercury levels, nutrient additions would not be useful in mitigating the mercury problem on the CRD. Primary productivity in these turbid lakes is generally light limited (Hecky and Guildford 1984; Ramsey et al. 1991), so nutrient additions will not have a significant impact on primary productivity or fish growth.

6.4 Liming

The liming of lakes through the addition of limestone has been proposed as a solution to mercury problems related to lake acidification. Although liming has been shown to cause some reduction of fish mercury levels in acidified lakes in Sweden, that decrease is small, on the order of 0.3 $\mu g\ g^{-1}$ for a 1 kg pike with a pre-liming mercury concentration of 1.5 $\mu g\ g^{-1}$ (Anderson and Borg 1990). There also is no reason to believe liming would be effective in northern Manitoba reservoirs because the mercury problems on the Rat-Burntwood and Nelson river systems were not caused by a pH reduction.

6.5 Inhibition of Methylation/Stimulation of Demethylation

One of the areas of research identified in the final report of the Canada-Manitoba Mercury Study Agreement was the investigation of methods to selectively stimulate the demethylating bacteria or to inhibit the methylators through the addition of a suitable substrate. This approach requires knowledge of the causative microorganisms and their metabolic pathways. A basic requirement is that the methylators and demethylators belong to different metabolic groups. The FEMP mercury studies included investigations to identify the major metabolic groups of bacteria which were responsible for methylation and demethylation in order to determine if the essential requirement of this approach was satisfied.

A pilot study in 1987 examined the role of methanogens in methylation and demethylation on one occasion during the routine summer sampling (Ramsey 1988a). The methanogens were found to account for the majority of methylation. Methanogen inhibition produced a 76% decrease in the methylation rate in flooded sediments. Demethylation rates increased with methanogen inhibition. This suggested the methylators and demethylators belonged to different metabolic groups, meeting the essential requirement for independent manipulation.

The results of the pilot study were sufficiently promising to warrant further investigation in the summer of 1988, when both methanogens and sulfate reducers were considered. The results of this more extensive investigation were disappointing. When the entire methylating season was considered, it was found that methanogens and sulfate reducers were the only important contributors to total methylation and demethylation in Methyl Bay flooded sediments and that the two groups were equally important in both processes (Table 1). Although a group may dominate one of the processes, but not the other, at a particular time and place, both groups make equal contributions over the season on average. Consequently, it may not be a simple matter to inhibit methylation or stimulate demethylation without having the same effect on the competing process.

6.6 Use Fish Species or Size-Classes with Lower Mercury Levels

It has been known for some time that the different fish species accumulate mercury to varying levels due to differences in feeding habits. Knowledge of these differences has been used to develop public information programs encouraging the domestic consumption of species with lower mercury concentrations. Unfortunately, the species with the lowest mercury levels, usually whitefish and suckers, also have the lowest commercial value. The knowledge that mercury level is dependent upon fish size might be equally useful in the rehabilitation of commercial fisheries based on the higher value predatory species. Small-mesh nets could be used to catch the smaller individuals in the populations of predatory species in lakes where mercury levels in these smaller size classes are acceptable for commercial marketing. Use of a "slot-fishery" was suggested by Jones et al (1985) and came up in discussion with Wayne Wysocki (Symbion Consultants, Winnipeg, MB, pers. comm.) as well. These smaller individuals also are more marketable than larger fish, further contributing to the viability of a commercial fishery. The resumption of commercial fishing in closed lakes might also have some effect in reducing fish mercury levels, as outlined in Section 6.3.1.

Although such a practise normally would not be acceptable to fisheries managers, potentially having a negative impact on future spawning stocks, it might be worthy of consideration in situations where there is very little to lose. One such lake is Wapisu, where standardized mercury levels in walleye and pike of 1.58 and 1.69 $\mu g\ g^{-1}$, respectively, were reported in 1985 (Green 1986). Based on the predicted rate of decline in M/D (Section 4.3; Figures 30 and 31), more than 60 years will be required for the mean mercury levels in these species to drop below the Canadian marketing limit. A "slot" commercial fishery may permit the immediate commercial utilization of this lake and present an alternative to the current practise of the subsidized fishing of cutter lakes. This should be attractive both to the fishermen and the subsidizing agency. Should it prove unviable as a result of stock depletion, then there would still be plenty of time for the stocks to recover as mercury levels in fish naturally decline.

7.0 Prevention of Mercury Problems in New Reservoirs

The current understanding of the cause of the reservoir mercury problem suggests a number of actions which might minimize or prevent the development of a mercury problem in a new reservoir. The dependence of elevated methylation rates on flooded organic matter indicates that some means of removing organic soils and vegetation before flooding, or at least reducing their organic content, should be effective. At this time, the only approach which can be recommended is the complete removal of organic matter from a basin before flooding (Hecky et al. 1987b). This measure obviously carries a significant cost and may not be practical for all but the smallest reservoirs. The partial clearing of a reservoir basin might be more feasible, but there is no means of predicting the reduction in peak fish mercury levels which can be expected for a given proportion of clearing.

Another approach which should be examined is the selective clearing of organic materials. Knowledge of which materials are most stimulatory of methylation, or the longest contributors to elevated methylation, would permit the concentration of basin preparation efforts in areas which would provide the greatest benefit. This measure requires information on the ability of specific types of organic matter to stimulate net methylation. Such information is not yet available but could be obtained from limnocorral experiments or field studies comparing the microbial methylation balance among sites with different vegetation cover.

Controlled burning of a reservoir basin prior to flooding might also be used to reduce organic loading, and therefore methyl mercury production. However, combustion may have some undesirable side-effects which remain to be identified.

The covering of bogs and other organic soils with inorganic soils might be used in specific circumstances where vehicle access is possible and there is a ready supply of material. An important technical consideration in this method is the means by which the inorganic cover can be maintained during reservoir filling. Some erosion should be expected as the cover is exposed to wave energy, and it remains to be determined how thick the cover must be to ensure sufficient cover material remains to isolate the underlying organics. This method would probably be inappropriate for use in the drawdown zone or at shallow depths within the wave zone.

It should be emphasized that none of these approaches has been tested in a new reservoir and, therefore, there is no information on which to base a cost-benefit analysis. Given the high profile of environmental mercury problems, their predicted longevity, and the lack of proven mitigation measures, hydroelectric developers should consider researching these prevention options further.

8.0 Prediction of Fish Mercury Levels in New Reservoirs

The prominence of the reservoir mercury problem has stimulated several attempts to develop a means of predicting fish mercury levels in new reservoirs, based on their physical and chemical characteristics, as an aid in assessment of their impact. The extensive fish mercury database and diverse physical features of the impoundments on the Churchill River diversion and Nelson River have made this area the focus of many modelling exercises.

Jones et al. (1985) examined the relationship between mercury concentration in fish and 28 physical and chemical characteristics of 17 reservoirs in Manitoba, Ontario, Quebec, and the Maritimes. They concluded that there was no means of predicting, with any measure of confidence, the occurrence or severity of mercury problems in new reservoirs based on existing knowledge. A number of variables were found to account for significant components of the variance in fish mercury, but too little of the total variance in fish mercury could be accounted for.

The study of Jones et al. (1985) failed to control a number of sources of variation which may have obscured any underlying relationships. The influence of upstream reservoirs was ignored and the temporal trend of fish mercury levels over time also was not controlled. Data for some reservoirs represented near peak values, while samples from other reservoirs were taken near the time of recovery (e.g., Cedar Lake in Manitoba).

Derksen and Green (1987) examined the relationship between mercury levels in fish and the extent of flooding (i.e., the % increase in lake area due to flooding) in lakes on the Churchill River diversion and Nelson River. They failed to find a significant, quantifiable, relationship between the variables which could be used for prediction. Their analysis indicated a positive correlation between the ranks of mercury in fish and extent of flooding, which suggested a cause-effect relationship, but this correlation was of no predictive use. The lack of a direct relationship between extent of flooding and mercury concentration was attributed to sampling variability and the number and complexity of factors which might influence such a relationship.

The efforts of Jones et al. (1985) and Derksen and Green (1987) concentrated on examination of the relationship between mercury levels in fish and within-lake physical and chemical characteristics. Subsequent studies have found the age of the reservoir at the time of sampling, and the influence of upstream reservoirs also must be considered.

Ramsey (1987b) found a significant correlation between mercury concentration in walleye and extent of flooding for lakes on the CRD and Nelson River by incorporating a qualitative measure of the influence of upstream reservoirs. Lakes were divided into 2 groups, those with, and those without a reservoir immediately upstream, and the groups were considered separately (Figure 33). The same approach was unsuccessful for northern pike, probably because the model considered upstream effects as a constant rather than as a

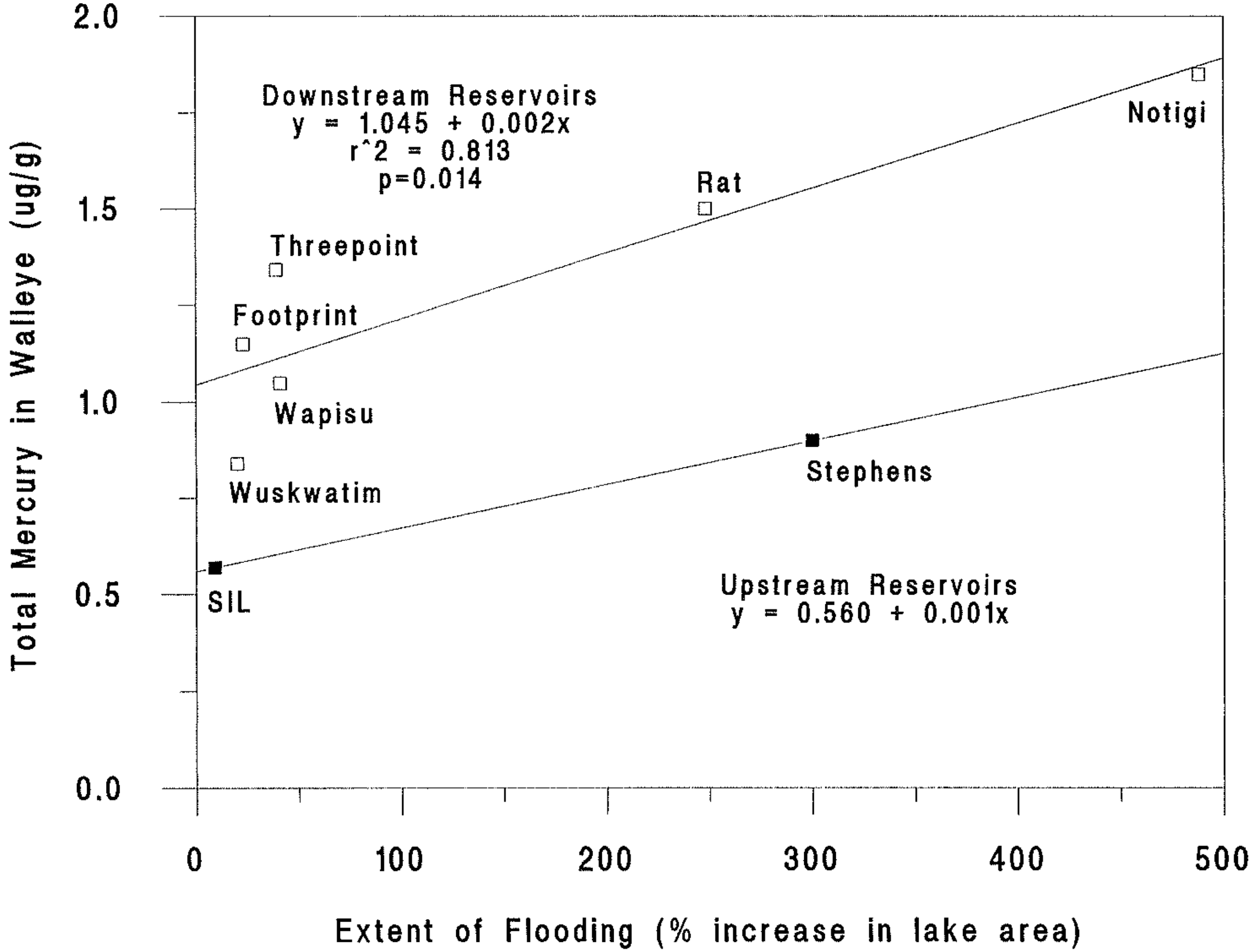

Figure 33. Relationship between extent of flooding (% increase in lake area due to flooding) and mercury concentration in walleye in lakes on the Churchill River diversion and Nelson River, Manitoba. Fish mercury data are means of all lake survey data collected between 1980 and 1984 as reported in Bodaly et al. (1984) and by Department of Fisheries and Oceans, Inspection Services, Winnipeg, MB. Extent of flooding values for all lakes from G. McCullough (Department of Fisheries and Oceans, Winnipeg, MB, pers. comm.). Figure modified from Ramsey (1987b) to reflect the latest (3 February 1992) extent of flooding estimates.

variable.

The importance of upstream reservoirs has since been incorporated in a more general model by Johnston et al. (1991). Once again working with data from the Churchill River diversion and the Nelson River, they found significant positive correlations between the peak post-impoundment, standardized, mercury burden and the rise in water level (RISE), percent flooding (PF), upstream percent flooding (UPF), flooded area to volume ratio (AVR), and the upstream flooded area to volume ratio (UAVR) for whitefish, walleye, and northern pike, the three species examined. Variables representing in-lake changes (RISE, PF, and AVR) accounted for 34-59% of the variation in mercury burden, while upstream changes accounted for 66-76% of the variation. Combining in-lake and upstream effects in a two-variable model accounted for 69-84% of the variation in mercury burden, significantly improving the models for whitefish and walleye but not for northern pike. Models using flooded area to volume ratio variables generally accounted for more of the variation in walleye and pike mercury burden than did models based on percent flooding variables.

The models performed well for flooded lakes on the Churchill River diversion but not in other reservoir systems. The model overestimated peak mercury burden in fish from Cookson Reservoir, Saskatchewan. This was attributed to differences in soils, vegetation, and water chemistry and the potential effect of these variables on microbial methylation. The models also performed poorly for reservoirs in Labrador and northern Quebec. This was considered to be a result of the dominant influence of in-lake effects in these projects. Additional investigation is required to determine the circumstances under which in-lake effects dominate over upstream effects, and vice-versa.

Despite the success of the models in accounting for variation in fish mercury levels among reservoirs on the CRD, it should not be assumed that the models are applicable to all northern Manitoba reservoirs. The models will probably perform well for the proposed reservoirs on the Burntwood River, but may not be as useful for the projects currently planned or under construction on the Nelson River with their different configuration. For example, there has been reference in past studies to the relationship between the elevation in fish mercury level and the rise in water level (e.g., Jackson 1987). Johnston et al. (1991) found this parameter accounted for a significant component of the variation in mercury level on the CRD, but rise in level should not be expected to be important in all circumstances. In the run-of-the-river reservoirs currently in operation on the Nelson River (e.g., Long Spruce and Limestone), significant increases in water level are largely contained within the natural banks of the river, with very little terrestrial organic matter being flooded. If no terrestrial organics are flooded, then the rise in water level should be of no consequence. Upstream flooding should have the dominant influence on methyl mercury availability in the reservoirs on the lower Nelson River.

8.1 Cross Lake Weir

There has been some uncertainty expressed by residents of the Cross Lake community about the potential impact of the proposed Cross Lake weir on mercury levels in fish (L. McKerness, Environment Canada, Winnipeg, MB, pers. comm.). This uncertainty seems related to the stated relationship between mercury in fish and rise in water level on the CRD. It is unlikely the resulting higher water level on Cross Lake will cause any long-term increase in fish mercury levels because the rise is contained within the natural lake basin. The weir is designed to raise the low water level without affecting the high water level. Therefore, there will be no backflooding of bog or forest terrain associated with the weir, unlike the situation along the CRD.

Experience in Sipiwesk Lake (Section 4.4) raises the question of the extent to which inundation of the macrophytes and sedges which have become established in the drawdown zone of Cross Lake will affect mercury levels in fish. Again, an elevation in fish mercury levels seems unlikely, due to important differences in shoreline composition between the lakes. The majority (92%) of the Cross Lake shoreline is bedrock controlled (Water Resources Branch 1974), which greatly limits the potential for revegetation of the drawdown zone during periods of low water. In Sipiwesk Lake, only 5% of the post-impoundment shoreline is bedrock-controlled, with the remainder comprised of flooded forest and wetland soils (Water Resources Branch 1974) which provide excellent substrate for growth of vegetation during periods of low water. Although Cross lake levels have fluctuated substantially since the Jenpeg Generating Station began operating in 1976, by as much as 3 m in a single year (Water Survey of Canada 1990), mercury levels in pike and walleye from Cross Lake are not higher than the average for lakes in northern Manitoba (0.3-0.4 $\mu g\ g^{-1}$) (Ramsey 1991). Any vegetation which becomes established in the drawdown zone apparently is not of sufficient quantity to cause a substantial elevation in net microbial methylation when it is flooded. Thus, construction of the weir should not cause increased fish mercury levels. This assessment should be confirmed by monitoring for at least 2 years after the weir is constructed.

9.0 Further Studies

The past five years of research on the mercury problem in the NFA area have answered many questions on the factors regulating microbial methylation and demethylation in new reservoirs. Our improved understanding of these processes has been applied in the development of useful models for prediction of the severity and duration of fish mercury problems in new reservoirs. This research program also has identified several questions which are in need of further investigation. These uncertainties are summarized below along with studies which might address these questions.

There is considerable circumstantial evidence which suggests the Kelsey Generating Station on the Nelson River is contributing to the mercury burden in Split Lake fish. It cannot yet be determined if this is in fact the case nor is the mechanism certain. The most probable causes are methyl mercury export from the upstream reservoir or downstream migration of contaminated fish. Another possibility is that fish in Split Lake may be feeding on fish killed in passing through the Kelsey Dam turbines.

The potential for methyl mercury export can be examined through measurements of methyl mercury in water along the Nelson River between Lake Winnipeg and Split Lake and in the other major inflows to the lake (i.e., the Aiken and Burntwood rivers). These data would indicate the location of any site of elevated methyl mercury production and determine if the methyl mercury is entering Spit Lake. The importance of fish movements from Sipiwesk Lake and the Kelsey forebay could be documented using a tagging program, as has been employed at Missi Falls on SIL. If neither of these studies finds a methyl mercury source which can explain the fish mercury levels in Split Lake, then the potential effect of fish killed in the Kelsey turbines, or the existence of a site of elevated net methylation within Split Lake should be investigated.

The numerous examples of elevated fish mercury concentrations downstream of reservoirs indicate the importance of considering downstream effects in assessments of the environmental impact of all new reservoirs. Methyl mercury can move downstream both in water and in fish. In some cases, the primary mechanism of transport, fish or water, is clear but in others there is no means of discriminating between these sources. The occurrence of much higher fish mercury levels in lakes immediately downstream from Notigi Reservoir than expected based on the extent of flooding is clear evidence of downstream transport of methyl mercury. Both mechanisms of transport probably are acting, but the relative importance of these sources remains to be determined. Fish movements out of Notigi Reservoir should be quantified to determine if they are making a significant contribution to downstream fish stocks, thereby increasing the mean mercury concentration in fish. Exports of methyl mercury in water also remain to be quantified as do the downstream extent of this transport and the mechanism by which any methyl mercury is lost from the water column. The methyl mercury survey should focus on the section of the CRD between Notigi Reservoir and Thompson as previous work indicated any methyl mercury leaving the reservoir is lost from the water column within this reach.

Studies to date have suggested that microbial demethylation rates are regulated by organic matter availability and total microbial activity. These observations need to be confirmed. There also is a need to determine why demethylation appears to be independent of sediment oxygen conditions, while methylation increases under anoxic conditions. This may provide at least a partial explanation for the differential stimulation of the methylating bacteria in flooded reservoir sediments. It is this greater stimulation of the methylators which is the cause of the elevated mercury levels in fish from reservoirs.

Concerns about the potential for elevated mercury concentrations in other food species (e.g., gull eggs, mergansers, bear), and their contribution to the mercury burden and potential health effects in humans, have been raised and need to be addressed. A pilot study should first be conducted to identify all other wild food sources, to evaluate the potential for elevated mercury levels in these sources, and to obtain an initial measure of mercury concentrations in those species for which there is the potential for elevated levels related to hydroelectric development. One criterion which might be used for selection of study species is to include all species for which fish forms part of the diet. Mercury analyses should be performed on all parts of the animals that are consumed by humans. These data can be used, along with an estimate of consumption, to determine if the animals represent a significant mercury source to consumers which may have been elevated due to flooding, and if a more extensive sampling program is warranted. This information also would be useful in assessing the potential for physiological or behavioral effects of any elevated mercury levels in these other animal groups.

There now appears to be sufficient knowledge of the cause of mercury problems in northern reservoirs to identify potential measures for the minimization or prevention of elevated fish mercury levels in new reservoirs and for the rehabilitation of fisheries in established reservoirs which continue to support elevated mercury levels. Among the proposed preventive measures, controlled burning requires an evaluation of effectiveness and an assessmegw of adverse environmental side-effects. Preliminary investigations could employ limnocorrals followed by field trials in a new reservoir if the initial results appear favourable.

Selective removal of organic matter requires knowledge on the ability of various terrestrial organic materials (plant species, soil types) to stimulate microbial methylation and the duration of this stimulation. This information could be obtained through methylation studies at selected reservoir sites with known vegetation/organic matter composition and known duration since impoundment. Unfortunately, vegetation maps were not prepared for most existing reservoirs, although a notable exception is the Complexe la Grande in northern Quebec which was extensively mapped prior to flooding.

Measures to rehabilitate fisheries which have been closed due to elevated mercury levels which might be employed on the CRD include intensive fishing, for the general reduction of mercury levels in fish, and selective fishing, to make use of species or particularly size-classes with lower mercury levels. Additional investigation is required to assess the benefits, costs, and long-term impacts on fish resources of these methods and their overall utility in the CRD.

The accumulated knowledge on the cause of reservoir mercury problems in northern Manitoba is now sufficient for the development of useful numerical models for the prediction of the severity of fish mercury problems in new reservoirs proposed for the CRD. Generalization of these models for application to projects other regions requires knowledge on the circumstances which determine the relative importance of in-reservoir and upstream effects. This requires information on the downstream extent of methyl mercury export from reservoirs and the factors affecting this distance, and some means of determining whether an upstream reservoir will have a diluting or contributing effect on methyl mercury availability in the reservoir in question. Extension of the model to other biophysical regions requires re-examination of the fish mercury/extent of flooding relationships for the relevant species. Specific field studies are required to determine the downstream extent of methyl mercury export, but the remaining issues can probably be examined in a modelling exercise using existing information.

An important part of impact assessment is the conduct of post-development studies to verify the precision and accuracy of the projection. In this regard, two seasons of post-construction monitoring of fish mercury levels in Cross Lake, using sampling methods consistent with those employed in the EMP, are suggested to confirm the prediction that the proposed weir for regulation of water levels on Cross Lake will not cause elevated fish mercury levels.

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11.0 Glossary of Technical Terms and Units of Measure

aerobic living, acting, or occurring only in the presence of oxygen.

amino acid organic compound containing both basic amino (NH_2) and acidic carboxyl (COOH) groups. Fundamental constituents of living matter, hundreds or thousands of amino acids are required to make each protein molecule.

anaerobes organisms living or occurring in the absence of free oxygen.

bacteria group of single or multi-celled, microscopic, prokaryotic organisms, lacking chlorophyll.

bioaccumulation the development of a higher concentration of a substance in an organism than in its surrounding environment.

carbanion an organic ion carrying a negative charge at a carbon position.

centrifuge a machine using centrifugal (spinning) force for separating substances of different densities or for removing moisture.

CH_3^- a methyl group which is composed of 1 carbon and 3 hydrogen atoms.

$^{14}CH_3HgI$ methyl mercuric iodide in which the carbon atom has been replaced with carbon-14, a radioactive carbon isotope.

CMMA Canada-Manitoba Agreement on the Study and Monitoring of Mercury in the Churchill River Diversion.

co-enzyme an organic compound which combines with an enzyme and plays an essential part in its catalytic reaction without being consumed in the process.

correlation the study of the simultaneous variation of two or more variables. A positive correlation is one in which one variable increases as the other increases. A negative correlation is one in which one variable decreases as the other increases.

demethylation process by which a methyl group is removed from a compound. In methyl mercury demethylation, methyl mercury is converted into elemental mercury.

DIC dissolved inorganic carbon.

electrophilic involving or having an affinity for electrons; electron-seeking.

eutrophic a type of lake with high productivity and high nutrient supply.

EMP Ecological Monitoring Program (provincial and Manitoba Hydro).

enzyme a protein which promotes chemical reactions in organisms but is not consumed by the reaction. Functions to accelerate a reaction or to permit a reaction to occur under conditions in which it would not normally proceed. Most enzymes are specific to one reaction.

FEMP Federal Ecological Monitoring Program.

Hg chemical symbol for mercury.

Hg^{2+} chemical notation for the mercuric ion.

$^{203}Hg^{2+}$ chemical notation for mercury-203, a radioisotope of mercury, occurring as a mercuric ion.

Hg° chemical notation for elemental mercury, which is a silver liquid at room temperature. This is the form of mercury used in thermometers.

in situ in the natural or original environment.

ion an atom or group of atoms which carries a positive or negative charge as a result of having lost or gained one or more electrons.

isotope any of two or more species of atoms of a chemical element with the same atomic number and position in the periodic table and nearly identical chemical behaviour but with differing atomic mass or mass number and different physical properties.

isotopic equilibrium the state in which in which all isotopes of an element are completely mixed.

km kilometre (1000 metres).

limnocorral large enclosures used to isolate a water mass from a larger body of water.

loss on ignition percentage of sample weight lost after all the organic matter is combusted. A measure of organic content.

LWCN Lake Winnipeg, Churchill and Nelson Rivers.

m metre.

mercuric (ion) a divalent mercury atom with a double positive charge.

mercuric reductase an enzyme which is necessary for the conversion (reduction) of mercuric ions to elemental mercury.

mercury a naturally occurring metallic element.

methane a colourless odourless flammable gaseous hydrocarbon (CH_4) that is a product of the decomposition of organic matter.

methanogen a type of bacteria capable of forming **methane** (i.e., is methanogenic) under **anaerobic** conditions.

methanogenesis the process of producing methane.

methyl mercuric ion (CH_3Hg^+) a positively charged methyl mercury molecule.

methyl mercury term used to identify a mercury atom to which one or more methyl (CH_3) groups are attached.

methylation process by which a methyl group is attached to an atom or molecule. In mercury methylation, mercuric ions are converted to methyl mercury.

methylcobalamin chemical name for Vitamin B_{12}.

mg milligram (10^{-3} gram).

mg L^{-1} milligrams per litre.

μg L^{-1} micrograms per litre.

μg g^{-1} micrograms per gram.

microbe minute (microscopic) organism.

microbial pertaining to **microbes**.

mm millimetre (10^{-3} metre).

M/D the ratio of specific methylation to specific demethylation rates. Also called the **methylation balance**.

methylation balance M/D ratio. A measure of the net rate of methyl mercury production by bacteria.

monomethyl mercury an organic form of mercury containing one methyl (CH_3) group.

$NADP^+$ nicotinamide adenine dinucleotide phosphate (oxidized form). Also called Co-enzyme II. An oxidizing-reducing co-enzyme involved in synthesis of metabolic products which is not consumed in the process.

NADPH nicotinamide adenine dinucleotide phosphate (reduced form).

net-plankton **plankton** organisms large enough to be captured in a mesh net of a specified size.

NFA Northern Flood Agreement.

ng nanogram (10^{-9} gram).

ng L^{-1} nanograms per litre.

ng g^{-1} nanograms per gram.

non-enzymatic not requiring an **enzyme**.

obligate biologically essential for survival.

oligotrophic a type of lake with low productivity and low nutrient supply.

organomercurial lyase an enzyme which is necessary for the removal of the methyl group from methyl mercury

photolysis chemical decomposition by the action of radiant energy (e.g., sunlight).

plankton small animals and plants which float or drift in lakes.

ppm a concentration expressed in parts per million. Equivalent units are $\mu g\ g^{-1}$ and $mg\ L^{-1}$.

primary productivity the total amount of organic matter synthesized by the autotrophic (mainly plants) organisms of na ecosystem.

protein a complex organic compound composed of many **amino acids**.

radioactive the property of having atoms that break up spontaneously and send out radiation.

radioisotope a **radioactive** isotope (form) of an element.

SIL Southern Indian Lake

sulphate reducers a group of bacteria which convert sulphate (SO_4) to hydrogen sulphide (H_2S; a gas which smells like rotten eggs) under anoxic conditions.

sulfhydryl group a chemical functional (SH) group, comprised of one sulphur and one hydrogen atom, occurring in the **amino acid** cysteine.

supernatant the liquid overlying material deposited by centrifugation.

Vitamin B_{12} a cobalt-containing vitamin required by many organisms.

EFFECTS OF THE CHURCHILL RIVER DIVERSION ON WATERFOWL USE OF THE RAT-BURNTWOOD RIVERS SYSTEM AND ON THE NELSON HOUSE BAND

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ABSTRACT

Surveys carried out in September 1988 indicated that use of the Rat-Burntwood rivers system by ducks in fall declined following completion of the Churchill River Diversion project. Duck numbers overall in 1988 were 95 percent lower than recorded during 1973 surveys. Dabbling ducks accounted for most of the decline (97 percent reduction), while diving duck numbers fell by 89 percent. Survey results were insufficient to assess levels of use by Canada geese with any degree of confidence.

The observed reduction in duck use of the Rat-Burntwood area cannot be attributed to a regional decline in populations, since no dramatic decline was evident in regional survey data (Stratum 24). It is more likely that the reduction was associated with a 4 to 5 m increase in water levels that removed marshy cover along shorelines, leaving behind poor quality habitat characterized by dead trees, fallen timber and other debris, and eroding banks. However, the limited number of survey years (1 pre- and 1 post-diversion) and the variability inherent in aerial waterfowl counts make it impossible to state, with certainty, the cause and significance of the changes observed.

Residents of Nelson House interviewed during the study indicated that low duck numbers, high water levels, and debris along flooded shores have reduced opportunities for Band members to hunt and view waterfowl.

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ACKNOWLEDGEMENT

We gratefully acknowledge the efforts of Peter Boothroyd, who conducted the field surveys, and hunter interviews, and who played a major role in preparing this report.

1.0 INTRODUCTION

In the early 1970's, Manitoba Hydro began to proceed with developments to harness the hydroelectric potential of the Nelson River. This involved the diversion of a major portion of the flow of the lower Churchill River into the Nelson River, referred to as the Churchill River Diversion (CRD) project, and regulation of the outflow from Lake Winnipeg through the Lake Winnipeg Regulation project. The CRD project is the subject of this report and is described in **3.0 CHURCHILL RIVER DIVERSION PROJECT.**

In response to concerns expressed about the effects of this development scheme on the natural environment and on northern communities that depended on natural resources, the governments of Canada and Manitoba initiated the Lake Winnipeg, Churchill and Nelson Rivers Study. In April 1975, a report was issued by the Lake Winnipeg, Churchill and Nelson Rivers Study Board (Study Board or LWCNRSB) which described the environmental and social impacts that were expected to occur as a result of the hydroelectric development. The report also recommended measures to minimize or offset impacts, and specified additional studies that should be carried out. Study Board findings and recommendations relevant to the present study are discussed in **4.0 PRE-PROJECT ENVIRONMENTAL ASSESSMENT.**

To ensure that all members of Indian communities affected by the hydroelectric development would be adequately compensated, the **Northern Flood Agreement (NFA)** was entered into on December 16, 1977 by Canada, Manitoba, Manitoba Hydro and the Northern Flood Committee. The Northern Flood Committee was established to represent the interests of the Nelson House, Norway House, Cross Lake, Split Lake and York Factory bands. Included in the **NFA** is the requirement that the adverse effects of the development be monitored to determine how Indian Band members have been affected and how they should be compensated.

The **NFA** also established an arbitration procedure whereby injured parties could file claims and obtain suitable remedies for damages suffered. One such claim, Claim 18, lead to the initiation of the Federal Ecological Monitoring Program (FEMP) in the spring of 1986. FEMP was to address federal responsibilities under the NFA with respect to environmental research and monitoring. Claim 18 and various provisions of the **NFA** are described in **APPENDIX A, NORTHERN FLOOD AGREEMENT**.

Funds provided through FEMP enabled the present study to be initiated in response to concerns expressed by Nelson House Band that opportunities to hunt waterfowl in the fall had declined following completion of the CRD project.

2.0 STUDY AREA

In the context of this study, the Rat-Burntwood rivers system extends from the western end of Wapisu Lake, located downstream from the Notigi Control Structure, to the southern end of Wuskwatim Lake (Figure 1). The study area includes Wapisu, Threepoint, Footprint, Wuskwatim and Honeymoon lakes, reaches of the Rat and Burntwood rivers connecting these lakes, and the section of the Burntwood River between Threepoint Lake and Hopover Rapids. The community of Nelson House is located on Footprint Lake. Notakikwaywin Lake, located approximately 15 km north of Nelson House and not connected to the Rat-Burntwood rivers system, was included in the study area for comparison purposes.

The topography of the area reflects the underlying bedrock and the area's geomorphic history, particularly that of glacial and post-glacial periods (LWCNRSB 1975a). The distribution of surface material has been influenced by glaciation and glacial Lake Agassiz. Organic deposits, glacial till, lacustrine clays and glaciofluvial sands comprise the surficial material. Permafrost is common, especially if surficial materials are fine and drainage is poor. Organic deposits are widespread, consisting mainly of various types of peat (LWCNRSB 1975b).

The shorelines of Wapisu, Threepoint and Wuskwatim lakes are bedrock-controlled to a large extent. Prior to the diversion, a thin fringe of sedge (*Carex* spp.) and willow (*Salix* spp.) occurred along the shore of Wapisu Lake. Marsh areas were found at the shallow end of embayments and at the outlet of the Lake. Threepoint Lake used to be quite shallow before the diversion, and pondweeds (*Potomogeton* spp.) were present throughout. Marshes occurred in small bays at the mouth of small tributaries; beaches were common. A wide low-lying area along the Burntwood River upstream of Wuskwatim Lake used to contain extensive marsh habitat prior to the diversion. Submerged aquatic plant growth was so dense in mid-summer that navigation was restricted to a narrow section of the central channel. Further upstream, the river flows through a series of rapids contained within elevated banks. Before diversion, about one-half of the shoreline of Wuskwatim Lake was bedrock controlled with low alluvium and marshy sites comprising the remainder. Extensive marsh habitat occurred at the south end of the Lake before diversion (LWCNRSB 1975a,b). Kellerhals Engineering Services

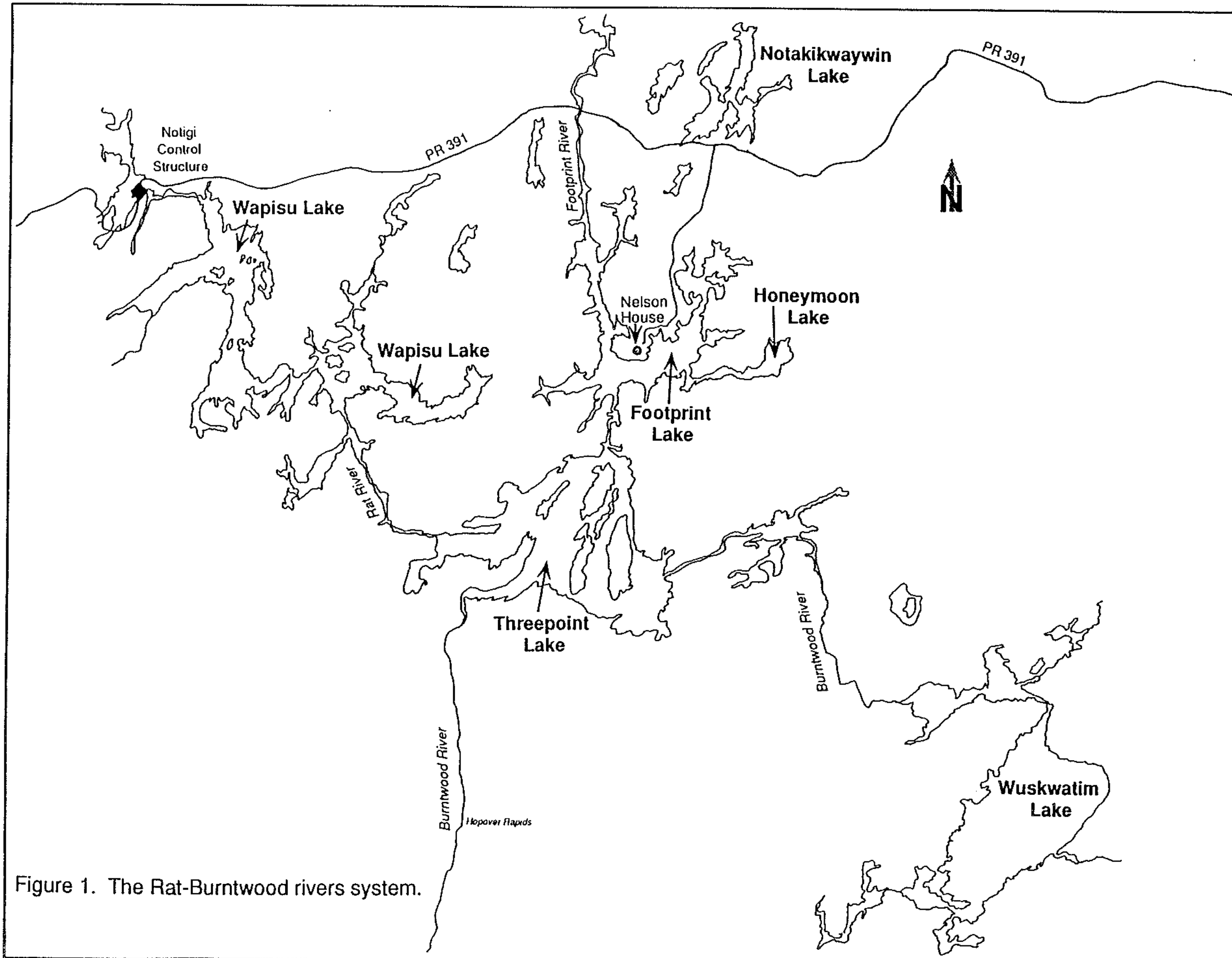


Figure 1. The Rat-Burntwood rivers system.

Ltd. (1988) noted that shorelines at the southern end of the lake are eroding at a rate of .2 m per year.

Black spruce (*Picea mariana*), white spruce (*Picea glauca*), trembling aspen (*Populus tremuloides)*, balsam poplar (*Populus balsamifera*) and white birch (*Betula papyrifera*) dominate most forest communities in the area. Willow, alder (*Alnus* spp.) and sedge communities occur on wet or saturated sites. Before diversion, marsh vegetation was characterized by a band of sedge along shoreline areas with an outer band of bulrush (*Scirpus* spp.) in deeper water. Floating aquatics, such as duckweed (*Lemna* spp.), were common in shallow sheltered areas, and pondweed (*Potomogeton* spp.) often occurred in deeper water beyond the bulrush margin (LWCNRSB 1975b). With the increased water levels associated with the diversion, marsh areas have become inundated so that shorelines today are obstructed by debris and dead standing trees. For example, Kellerhals Engineering Services Ltd. (1988) estimated that, at Footprint and Honeymoon lakes, the length of shoreline containing standing debris had increased from 0.2 km in 1972 to 56 km in 1985.

The climate is continental and is characterized by short, cool summers and long, cold winters. Annual precipitation averages 45 cm, 60 percent of which falls as rain between April and October (LWCNRSB 1975b).

3.0 CHURCHILL RIVER DIVERSION PROJECT

The following details of the CRD Project were extracted from the Summary Report of the Lake Winnipeg, Churchill and Nelson Rivers Study (LWCNRSB 1975c), supplemented by other information from Manitoba Hydro.

The Churchill River Diversion is a major component of hydroelectric development on the Nelson River. In simple terms, its purpose in the overall scheme is to divert approximately 75 percent of the flows of the lower Churchill River into the Nelson River drainage to augment Nelson River flows and maximize the production of electric power at generating stations located along that river.

A control structure was installed at Missi Falls where flows from Southern Indian Lake historically entered the lower Churchill River (Figure 2). The control structure permitted natural outflows to be held back in Southern Indian Lake and caused the mean level of the lake to increase by about three metres. The rise in water level enabled water to be diverted from Southern Indian Lake at South Bay and into the Rat River located in the Nelson River drainage basin. The water flows by gravity from South Bay to Issett Lake on the Rat River through an excavated channel. Control of the flow into the Rat-Burntwood rivers system is achieved by a control structure located at the outlet of Notigi Lake.

Construction of the Missi Falls and Notigi control structures began in February 1973 and September 1973, respectively. Both structures were completed by August 1976. Connection of Southern Indian Lake to the Rat River occurred on June 2, 1976, with breaching of the South Bay cofferdam. The diversion was phased in over 1976 and 1977.

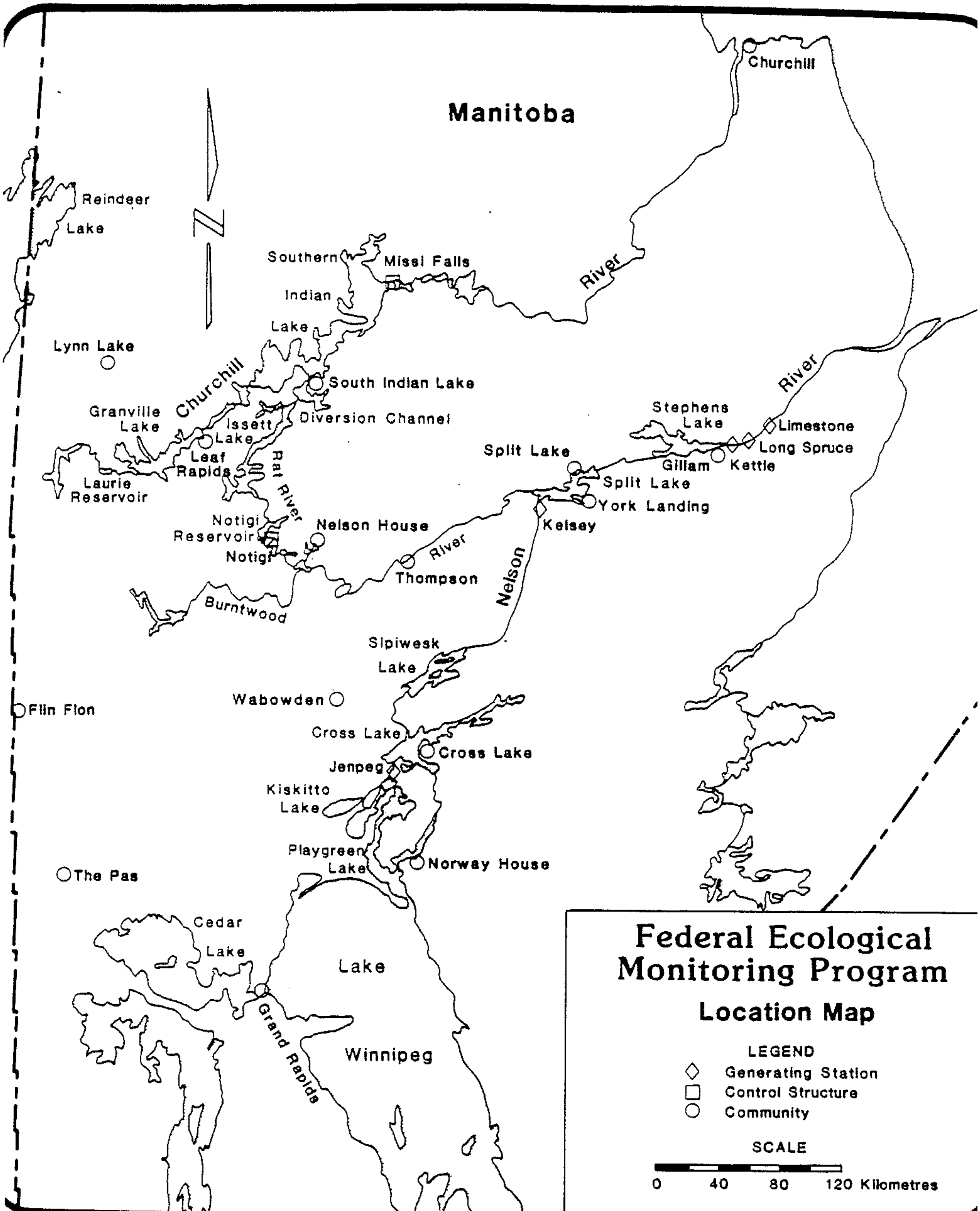


Figure 2. Area affected by the Churchill River Diversion Project.

4.0 PRE-PROJECT ENVIRONMENTAL ASSESSMENT

4.1 Study Findings

Following its investigation into the potential physical impacts of the Churchill River Diversion, the LWCNRSB (1975a) concluded the following:

> *"After the diversion of flows from the Churchill River and regulation at the Notigi control structure, the mean discharge in the Burntwood River will be increased nine to ten-fold. The impact on lakeshores underlain by bedrock will be one of erosion until bedrock is reached. Such shorelines will remain indefinitely cluttered with fallen trees. Embayments and other protection shorelines will remain congested with standing dead trees. Sand beaches and marshes will be inundated and will not redevelop to the extent they were in the natural state. Steep clay banks will continue to erode, although at an accelerated rate. These shorelines will likely appear much as they did in the natural state. Steep bedrock shorelines will not be significantly altered."*
>
> *"Each shoreline type will be affected differently. Little change will occur where bedrock controls the channel width but clay shores may recede approximately 60 to 70 ft. on either side. The main difference between riverbanks and lakeshores is the removal of flooded and fallen trees. Riverbanks will not likely contain the amount of dead standing trees that will typify protected lakeshores."*

Webb and Foster (1974) assessed the impact of the CRD project on waterfowl and waterfowl habitat. Concerning the effect of the diversion on fall waterfowl use of the Rat-Burntwood rivers system, the following predictions were made:

> *"Features which make certain areas along the Rat and Burntwood Rivers attractive to staging waterfowl will be lost after flooding; these include the large expanses of sedge and exposed mud flats. Food on the latter areas, both plant and animal, is the critical factor in attracting ducks and geese.*

Large expanses of potentially productive low land are being flooded throughout the Diversion route. The time required for aquatic vegetation to establish on these areas will determine future attractiveness to waterfowl. Clearing along shorelines in these areas would greatly speed reestablishment of attractive habitat.

Waterfowl using the large, open lake areas for shore migration stops will not be significantly affected by flooding."

4.2 Study Board Recommendations

The Study Board made a number of general recommendations that concerned the entire Nelson River hydroelectric development scheme, and others that were specific to certain issues or geographic areas.

Recommendations of general application that are relevant to the present study are:

5. *That a mechanism be established to deal with social and related economic issues including:*

   (c) *monitoring and analysis of ongoing social and economic changes related to hydroelectric development and more generally northern development.*

10. *That appropriate government departments and agencies develop and implement a long-term coordinated ecological monitoring and research program to allow impact evaluation and to assist in the ongoing management of the affected area.*

No specific recommendations were made in relation to the impacts of the CRD project on waterfowl use of the Rat-Burntwood area.

5.0 METHODS

5.1 Impact Assessment and Impact Management Approach

The flowchart in Figure 3 illustrates the approach used in determining how waterfowl-related interests and opportunities of Nelson House Band Indians in fall have been affected by the CRD project and, therefore, what compensation is required by the **NFA**. References are made in the flowchart to the pertinent recommendations of the Study Board and provisions of the **NFA** which are addressed by the process. Reference to "adverse" effects or impacts in this report reflects terminology of the **NFA** and does not represent any pre-conclusion of this study.

The assessment process begins with identifying the effects of the CRD project on waterfowl use of the Rat-Burntwood area in fall (biophysical impacts). The direct, indirect, or cumulative consequences of biophysical impacts are expected to affect the economic, cultural, and social life of Nelson House people.

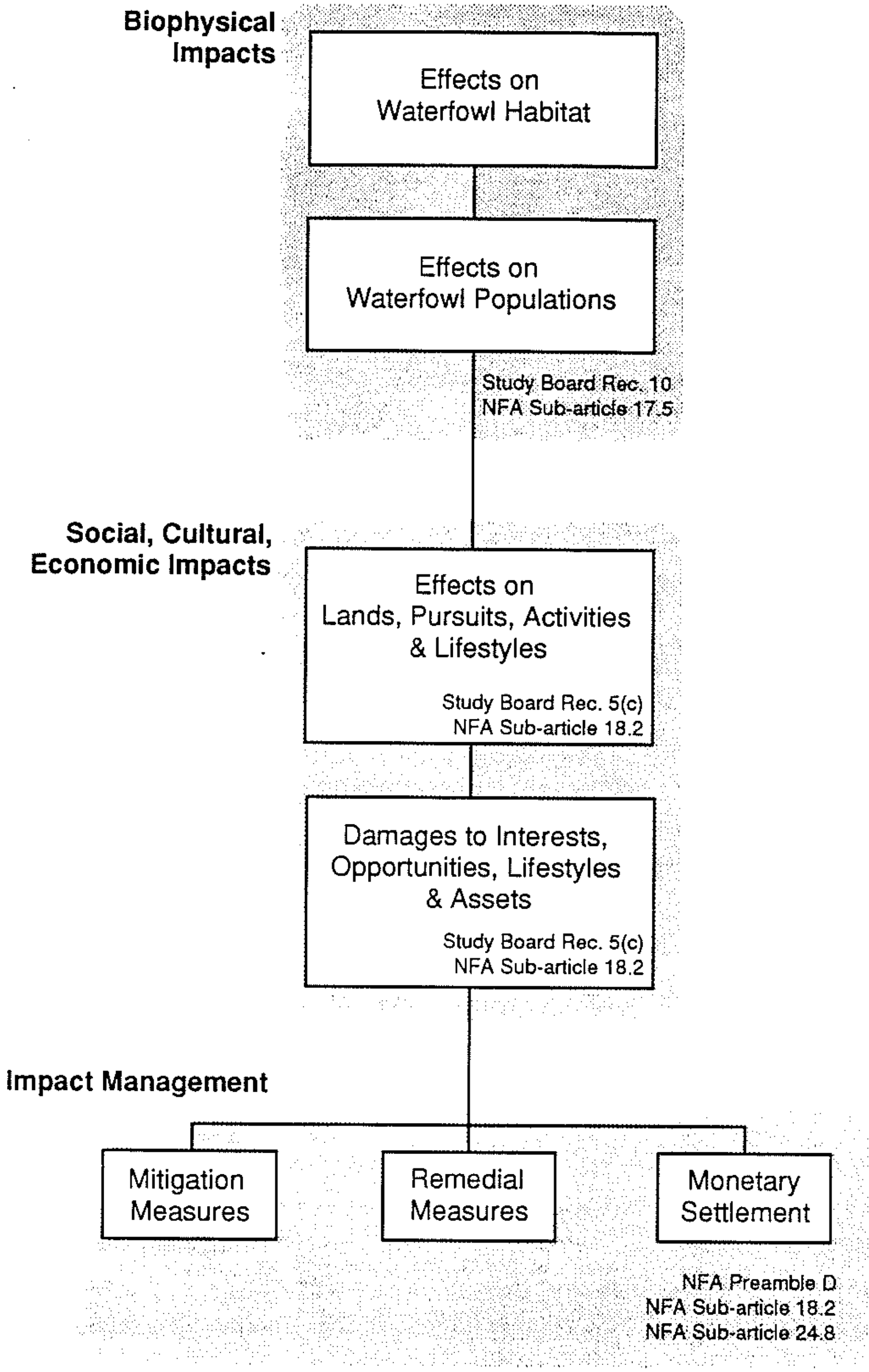


Figure 3. The impact assessment and impact management approach used in the study.

5.1.1 Biophysical Impacts

Determining how the CRD project has affected waterfowl use of the Rat-Burntwood area partially fulfills Recommendation 10 of the Study Board and **NFA** sub-article 17.5 (see Appendix A).

Recommendation 10 recognizes that ongoing management of the Rat-Burntwood area requires that information first be gathered on how the area has been impacted by development. The present study focuses on how use of the Rat-Burntwood area by fall-staging waterfowl has been affected by the CRD project.

NFA sub-article 17.5 requires that monitoring of effects of the CRD project be carried out to "provide such information as may be necessary to give effect" to the **NFA** (see Appendix A). Giving effect to the **NFA** means ensuring that all members of the five **NFA** bands, who have been or may be adversely affected by the hydroelectric development, are dealt with or compensated "fairly and equitably" (**NFA** Preamble D).

Collection of data that document the effects of CRD on waterfowl use of the Rat-Burntwood area constitutes compliance with sub-article 17.5 with respect to the waterfowl-related impacts that the Nelson House Band claims to have experienced. In addition, documentation of effects on waterfowl in the Rat-Burntwood area represents a response, in part, to Part III of Claim 18, which embodies the intent and meaning of Recommendation 10 and **NFA** sub-article 17.5. Claim 18 is described in Appendix A.

5.1.2 Socio-Economic-Cultural Impacts

Determining how CRD has affected the waterfowl-related interests of Nelson House Band people partially fulfills Study Board Recommendation 5(c) and **NFA** sub-article 18.2.

Recommendation 5(c) called for "monitoring and analysis of ongoing social and economic changes related to hydroelectric development". The present study addresses those changes experienced by Nelson House Band that are associated with effects on waterfowl use of the Rat-Burntwood area, i.e., changes in waterfowl hunting opportunities.

Collection of information that documents the effects of CRD on the waterfowl hunting opportunities of Nelson House Band constitutes compliance with **NFA** sub-article 18.2 with respect to the waterfowl-related impacts experienced by the Nelson House Band. In addition, this documentation represents a response, in part, to Part II of Claim 18 which embodies the intent and meaning of Recommendation 5(c) and **NFA** sub-article 18.2. Finally, information dealing with adverse effects on waterfowl hunting opportunities allows damages to the waterfowl-related interests, opportunities, lifestyles and assets of Nelson House Band Indians to be defined.

5.1.3 Impact Management

Impact Management refers to a mechanism, or set of mechanisms, by which the impacts of a project may be managed, whether they be biophysical (e.g., directly affecting waterfowl) or socio-economic-cultural (e.g., affecting waterfowl hunting opportunities). Suitable and adequate compensation for adverse effects on **NFA** bands would constitute redress for damages suffered by the bands. **NFA** sub-article 24.8 refers to three forms of compensation or redress for damages suffered: "mitigating measures, remedial measures and monetary settlement."

5.2 Data Collection

5.2.1 Effects on Waterfowl

Fall staging waterfowl use of the Rat-Burntwood area following diversion was determined through a series of low-altitude aerial surveys carried out in 1988. The surveys were conducted using a Bell 206B helicopter on September 1, 9 and 26. Altitudes of between 30 and 40 m above ground or water level were maintained during the surveys and the helicopter was positioned to enable the single observer to note the locations of waterfowl distributed along the shoreline.

Survey routes followed shorelines of Wapisu, Threepoint, Wuskatim and Honeymoon lakes and the Rat and Burntwood rivers (Figure 4). A total of 181.6 km were surveyed on September 1, 169.1 km on September 9, and 196.4 km on September 26 (Table 1). In addition, on September 26, 14.2 km of Notakikwaywin Lake shoreline were surveyed to determine whether the level of waterfowl use of areas located on the Rat-Burntwood rivers system and off the system were similar. Notakikwaywin Lake is located approximately 15 km north of Nelson House (Figure 4).

5.2.2 Effects on Nelson House Band

On February 19, 1988, one-on-one interviews were held with seven waterfowl hunters at Nelson House. Hunters were asked to describe their waterfowl hunting experiences before and after the Churchill River diversion, traditional hunting locations prior to diversion, hunting locations following the diversion, hunting success at traditional hunting locations before and after the diversion, and species of waterfowl hunted. They were also asked to share their observations on changes in numbers and distribution of waterfowl and their perceptions on the causes of these changes and the influence of the Churchill River Diversion. Because the purpose of the study was to investigate the effects of the diversion on fall waterfowl use and fall hunting, discussions focussed on the fall hunting period.

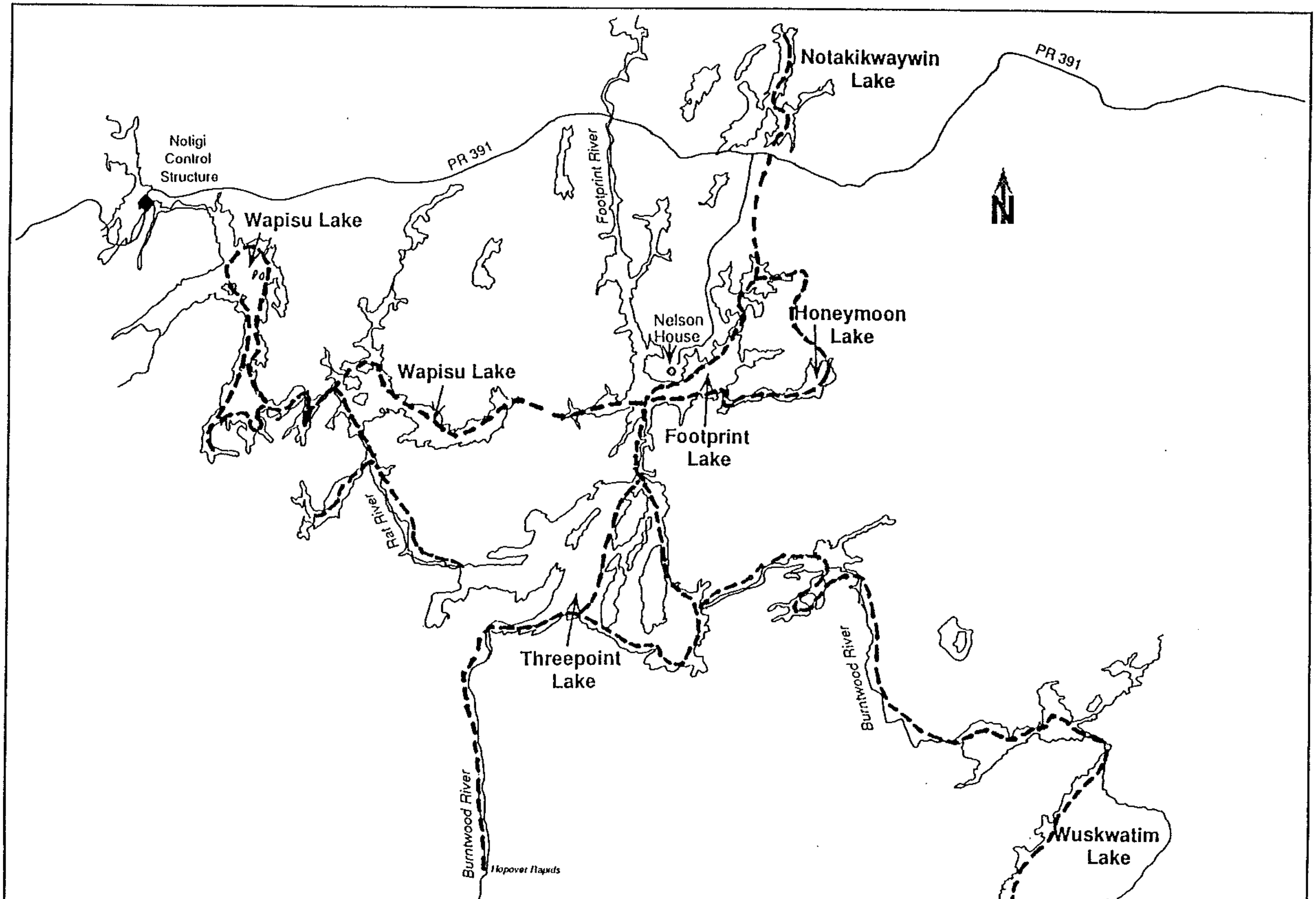


Figure 4. Rat-Burntwood river system: waterfowl survey route, fall 1988.

Table 1. Lengths (km) of post-diversion surveys of the Rat-Burntwood river system.

Area	Sept. 1, 1988	Sept. 9, 1988	Sept. 26, 1988
Wapisu Lake	59.2	38.4	58.6
Rat River	20.0	20.0	20.0
Burntwood River (upstream)	17.6	17.6	17.6
Threepoint Lake	9.0	9.0	16.1
Burntwood River (downstream)	32.2	32.2	32.2
Wuskwatim Lake	34.8	43.1	43.1
Honeymoon Lake	8.8	8.8	8.8
Notakikwaywin Lake	n.s.	n.s.	14.2
TOTAL*	**181.6**	**169.1**	**196.4**

* Does not include Notakikwaywin Lake.
n.s. = not surveyed

6.0 RESULTS

6.1 Waterfowl Use

Data on waterfowl use have been summarized for the following survey areas: Wapisu Lake, Rat River, Burntwood River upstream of Threepoint Lake, Threepoint Lake, Burntwood River downstream of Threepoint Lake, Wuskwatim Lake, Honeymoon Lake, and Notakikwaywin Lake (Figure 1). Where applicable, boundaries between areas were chosen to coincide with those of Webb and Foster's (1974) pre-diversion study to enable comparison of pre-and post-diversion counts. All survey areas except Notakikwaywin Lake are located along the Rat-Burntwood rivers system and are affected by the CRD project. Notakikwaywin Lake lies just north of Provincial Road 391, approximately 15 km northeast of Nelson House, and is not influenced by the CRD project.

Date presented here are summarized for dabbling ducks (Anatinae), diving ducks (primarily Aythyinae), and Canada Geese (Branta canadensis).

Highest densities of dabbling ducks observed per survey segment of the Rat-Burntwood system in fall 1988 (Figure 5, Table 2) ranged from 0.02 birds/km (Wuskwatim Lake) to 2.51 birds/km (Honeymoon Lake). No dabblers were seen on Threepoint Lake during any of the three surveys. The largest number of dabblers observed on a given survey was 106 birds (0.63 ducks/km) on September 9. The 2.47 dabblers/km recorded at Notakikwaywin Lake on September 26 was 7.3 times greater than the highest density (0.34 birds/km) observed on the Rat-Burntwood system on that date (Honeymoon Lake).

Maximum numbers of diving ducks ranged from 0.03 birds/km on the downstream segment of Burntwood River to 1.04 birds/km on Honeymoon Lake (Figure 6, Table 3). The largest number of divers observed on a single survey was 35 (0.21 birds/km) on September 9. Twenty-seven (27) divers (1.91/km) were recorded on Notakikwaywin Lake on September 26, the highest same-day density recorded on the Rat-Burntwood system was 0.23 diver/km (Honeymoon Lake).

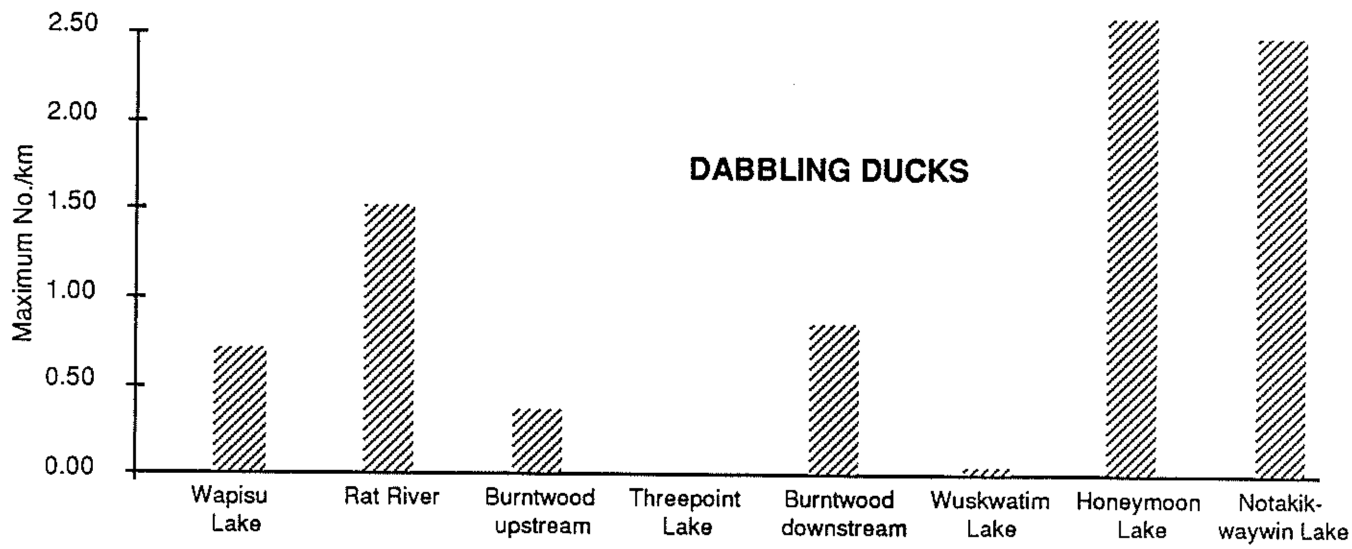


Figure 5. Dabbling duck use of the Rat-Burntwood area in Fall, 1988

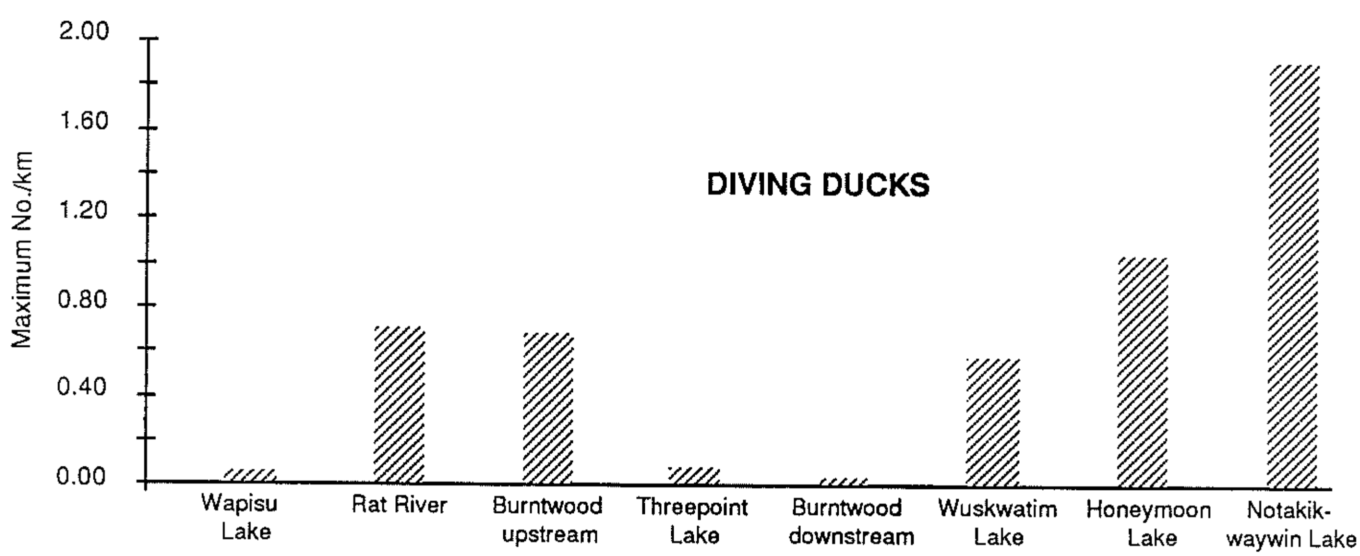


Figure 6. Diving duck use of the Rat-Burntwood area in Fall, 1988

Table 2. Numbers and densities* of dabbling ducks observed during post-diversion surveys of the Rat-Burntwood rivers area.

Area	1 Sept. 1988		9 Sept. 1988		26 Sept. 1988		Highest Count	Highest Density
	no.	density	no.	density	no.	density		
Wapisu Lake	0		25	0.65	10	0.17	25	0.65
Rat River	4	0.20	30	1.50	0		30	1.50
Burntwood River (upstream)	5	0.28	4	0.23	6	0.34	6	0.34
Threepoint Lake	0		0		0		0	0.00
Burntwood River (downstream)	1	0.03	25	0.78	0		25	0.78
Wuskwatim Lake	0		0		1	0.02	1	0.02
Honeymoon Lake	2	0.30	22	2.51	3	0.34	22	2.51
Notakikwaywin Lake	n.s.		n.s.		35	2.47	35	2.47
TOTAL & AVG. DENSITY**	**12**	**0.07^**	**106**	**0.63^**	**20**	**0.1^**	**144**	

* birds/km
** Excludes Notakikwaywin Lake
^ Based on total survey length: 181.6 km (1 Sept.); 169.1 km (9 Sept.); 196.4 km (26 Sept.)
n.s. = not surveyed

Table 3. Numbers and densities* of diving ducks observed during post-diversion surveys of the Rat-Burntwood rivers area.

Area	1 Sept. 1988		9 Sept. 1988		26 Sept. 1988		Highest Count	Highest Density
	no.	density	no.	density	no.	density		
Wapisu Lake	1	0.02	2	0.05	2	0.03	2	0.05
Rat River	0		14	0.70	0		14	0.70
Burntwood River (upstream)	0		12	0.68	0		12	0.68
Threepoint Lake	0		0		1	0.06	1	0.06
Burntwood River (downstream)	0		1	0.03	0		1	0.03
Wuskwatim Lake	20	0.58	5	0.12	5	0.12	20	0.58
Honeymoon Lake	7	1.04	1	0.11	2	0.23	7	1.04
Notakikwaywin Lake	n.s.		n.s.		27	1.91	27	1.91
TOTAL & AVG. DENSITY**	**28**	**0.15^**	**35**	**0.21^**	**10****	**0.05^**	**84**	

* birds/km
** Excludes Notakikwaywin Lake
^ Based on total survey length: 181.6 km (1 Sept.); 169.1 km (9 Sept.); 196.4 km (26 Sept.)
n.s. = not surveyed

Maximum numbers densities of dabblers, divers and unidentified ducks combined as shown in Figure 7 and Table 4, varied from 0.55/km (Threepoint Lake, September 1) to 2.63/km (Honeymoon Lake, September 9). The largest number of ducks (141; 0.83/km) was observed on September 9. The September 26 survey of Notakikwaywin Lake yielded 4.38 ducks/km, compared to the maximum of 0.57/km recorded on the same day in the Rat-Burntwood river system (Honeymoon Lake).

Canada geese were observed along only two survey segments: Rat River and Wuskwatim Lake (Figure 8, Table 5). Most of the 75 geese (1.74/km) were observed upstream of Wuskwatim Lake on September 9. No geese were observed at Notakikwaywin Lake.

In summary, low numbers of ducks were observed on the Rat-Burntwood system during post-diversion surveys. Densities were considerably higher during the single survey of Notakikwaywin Lake, which was not affected by the diversion. Canada goose numbers were low in all surveyed areas.

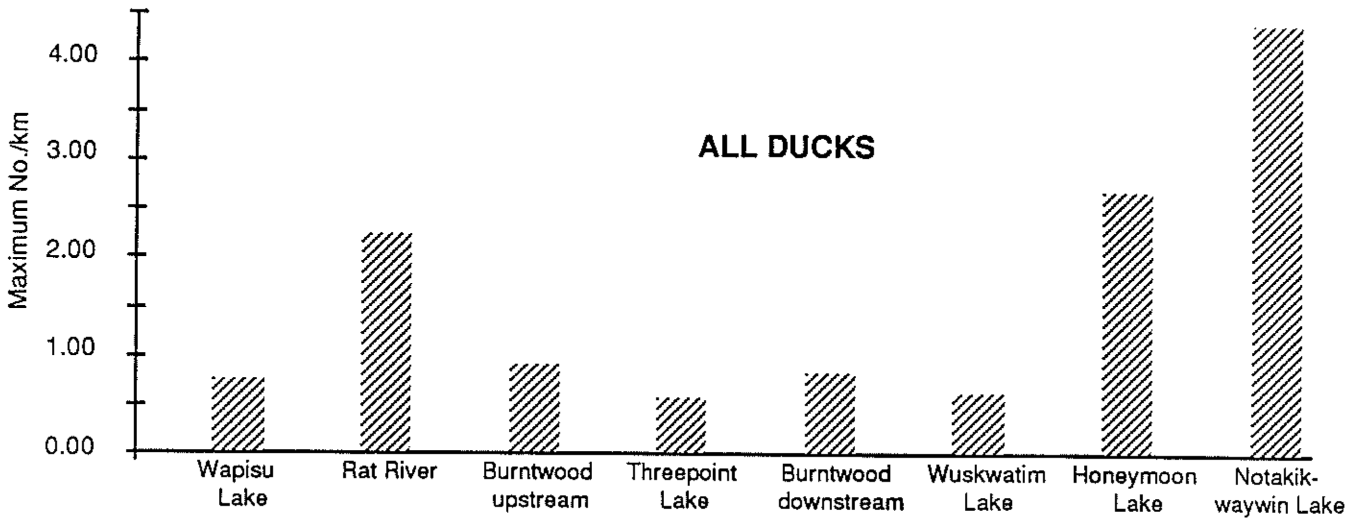


Figure 7. Duck use of the Rat-Burntwood area in Fall, 1988

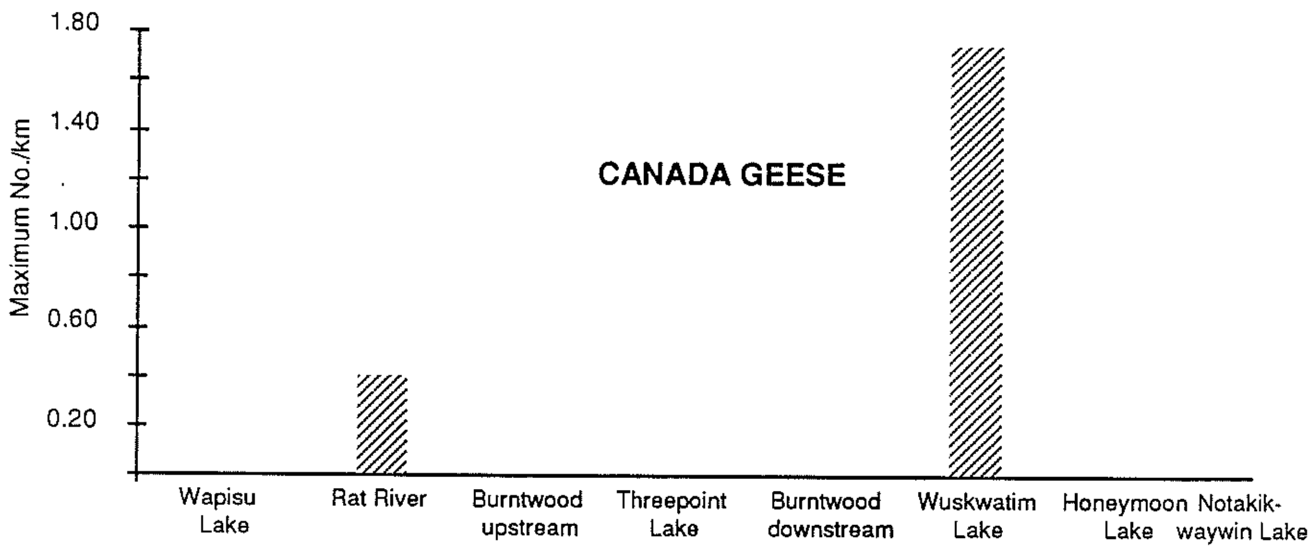


Figure 8. Canada goose use of the Rat-Burntwood area in Fall, 1988

Table 4. Numbers and densities* of ducks observed during post-diversion surveys of the Rat-Burntwood rivers area.

Area	1 Sept. 1988		9 Sept. 1988		26 Sept. 1988		Highest Count	Highest Density
	no.	density	no.	density	no.	density		
Wapisu Lake	1	0.02	27	0.70	12	0.20	27	0.70
Rat River	4	0.20	44	2.20	0		44	2.20
Burntwood River (upstream)	5	0.28	16	0.91	6	0.34	16	0.91
Threepoint Lake	5	0.55	0		1	0.06	5	0.55
Burntwood River (downstream)	1	0.03	26	0.81	0		26	0.81
Wuskwatim Lake	20	0.58	5	0.12	6	0.14	20	0.58
Honeymoon Lake	9	1.33	23	2.63	5	0.57	23	2.63
Notakikwaywin Lake	n.s.		n.s.		62	4.38	62	4.38
TOTAL & AVG. DENSITY**	**45**	**0.25^**	**141**	**0.83^**	**30****	**0.15^**	**223**	

* birds/km
** Excludes Notakikwaywin Lake
^ Based on total survey length: 181.6 km (1 Sept.); 169.1 km (9 Sept.); 196.4 km (26 Sept.)
n.s. = not surveyed

Table 5. Numbers and densities* of Canada geese observed during post-diversion surveys of the Rat-Burntwood rivers area.

Area	1 Sept. 1988		9 Sept. 1988		26 Sept. 1988		Highest Count	Highest Density
	no.	density	no.	density	no.	density		
Wapisu Lake	0		0		0		0	0.00
Rat River	8	0.40	2	0.10	0		8	0.40
Burntwood River (upstream)	0		0		0		0	0.00
Threepoint Lake	0		0		0		0	0.00
Burntwood River (downstream)	0		0		0		0	0.00
Wuskwatim Lake	0		75	1.74	0		75	1.74
Honeymoon Lake	0		0		0		0	0.00
Notakikwaywin Lake	n.s.		n.s.		0		0	0.00
TOTAL & AVG. DENSITY**	**8**	**0.04^**	**77**	**0.46^**	**0**	**0**	**83**	**0.46**

*birds/km
** Excludes Notakikwaywin Lake
^ Based on total survey length: 181.6 km (1 Sept.); 169.1 km (9 Sept.); 196.4 km (26 Sept.)

6.2 Hunter Interviews

Members of the Nelson House Band have traditionally hunted waterfowl during both spring and fall migrations. Waterfowl hunters expressed their concerns abut the effects of diversion during interviews conducted as part of this study (Figure 9). Hunters felt that the availability of ducks and geese at traditional hunting sites is considerably lower than it was before CRD. Hunters said that, before diversion, ducks and geese were plentiful throughout the area and hunters did not have to travel far to reach good hunting areas. They said they used to encounter numerous ducks along Rat River between Wapisu and Threepoint lakes and along the Burntwood River both above and below Threepoint Lake, and that upper reaches of Burntwood River in particular used to be frequented by young ducks. Ducks could usually be found in good numbers in shallower bays off Wapisu Lake, in the tall grass area that used to be located where Rat River entered Threepoint Lake, and in various locations in Threepoint Lake. Now, duck numbers are so low that hunters are forced to travel outside the Rat-Burntwood system to hunt waterfowl.

Hunters described at length how the increased water levels caused old shorelines to disappear and stands of trees to be inundated. Flooding of former shorelines has left fallen trees, logs and other debris in wide swaths along lake shores through much of the area. This has made boat navigation and shoreline access very difficult or impossible. Access problems have contributed to the reduction in opportunities to hunt waterfowl and moose. Hunters related how they would hear waterfowl or moose somewhere along the shoreline but, because of the debris, they could not move close enough to see or shoot animals. At times, even when it was possible to see ducks rise from dead trees standing along the shoreline, shooting the birds was pointless because birds often fell in areas where they could not be retrieved.

Extremely low numbers of waterfowl, combined with the difficulties of boat travel, have forced hunters to abandon their traditional hunting locations in the Rat-Burntwood area. Alternate hunting areas away from the diversion route are more difficult to get to and necessitate travelling greater distances resulting in increased travel costs.

WATERFOWL-RELATED CONCERNS AND ADVERSE EFFECTS

Observations of Nelson House Band Hunters*

Habitat

- used to be lots of grassy shallow-water areas in Threepoint Lake
- Honeymoon Creek used to be a narrow windy channel; now it's a lake
- old shorelines are gone; water is now into the trees

Waterfowl

- used to find young ducks along Burntwood River above Threepoint Lake
- used to see lots of ducks along Burntwood River downstream of Threepoint Lake, but not any more
- ducks used to be plentiful all along Rat River, but not any more
- bay at south end of Wapisu Lake was good for ducks and geese
- ducks and geese used to land in Threepoint Lake
- used to be lots of ducks in Wuskwatim Lake especially at south end

Hunting

- didn't have to travel far to reach good hunting areas
- debris in the water makes traveling by boat difficult
- traveling by boat at night is now too hazardous
- junk and debris along the shoreline makes ducks hard to see; when you do see them, you can't move close enough to them
- ducks that are shot cannot be retrieved due to trees and debris
- Threepoint Lake and Honeymoon Creek used to be good hunting areas
- used to be able to go to Honeymoon Lake and return with 10 ducks; now it's hard to see them let alone harvest them
- hunting along Rat River used to be good, but not any more
- ducks used to be plentiful in the back bays of Wapisu Lake
- creek entering northeast end of Footprint Lake used to be good; now you can't travel along it because of fallen trees
- used to be able to harvest ducks and geese on Threepoint Lake using blinds
- inland ponds are hard to get to; travel costs are greater

*Comments listed here represent the opinions of individual hunters

Figure 9. Waterfowl-related concerns and adverse effects identified during hunter interviews at Nelson House.

7.0 DISCUSSION

7.1 Pre- and Post-Diversion Waterfowl Use

Aerial surveys of shorelines in the Rat-Burntwood area were carried out before the Churchill River Diversion on September 7 and 26 in 1973 by Webb and Foster (1974) as part of the Lake Winnipeg, Churchill and Nelson Rivers Study. The September 7 survey was flown in a Cessna 337; a Cessna 180 was used for the September 26 survey. Survey altitudes ranged from 30 to 100 m and all bird counts were made within 150 m of shorelines. Honeymoon Lake, Notakikwaywin Lake and the reach of Burntwood River upstream of Threepoint Lake were not surveyed by Webb and Foster. Other water areas surveyed in 1973, in general, are comparable with those surveyed in 1988. Results of these surveys are summarized in Tables 6 through 9.

Comparison of 1973 and 1988 survey results for dabblers, divers, and all ducks combined (Tables 2-4 and 6-8) suggests that duck use along the Rat-Burntwood rivers system was considerably greater before diversion than after. Although more Canada geese were observed in 1973 than in 1988 (Table 9), their overall low numbers and spotty distribution make it difficult to draw any meaningful conclusions about the effects of CRD on fall staging populations.

Table 10 and Figures 10-14 show the percentage change in numbers of ducks that occurred between 1973 (pre-diversion) and 1988 (post-diversion) in the Rat-Burntwood area. The biggest changes in dabbler numbers occurred in Threepoint Lake (100 percent reduction) and Wuskwatim Lake (99.89 percent reduction) based on highest survey counts. The reduction in dabbler presence at the other three areas ranged from 85 to 95 percent. Numbers of divers were also lower in 1988 at Threepoint Lake (-96.8 percent), Wuskwatim Lake (-92.4 percent), Wapisu Lake (-90.9 percent), and lower Burntwood River (-80 percent); on Rat River, divers showed an increase (40 percent) in 1988, although counts there in both survey years were low.

Comparison survey results for dabblers, divers, and total ducks shows a substantial reduction in numbers from pre- to post-diversion periods over the Rat-Burntwood

Table 6. Numbers of dabbling ducks observed during pre-diversion surveys of the Rat-Burntwood rivers area.

Area	7 Sept. 1973	26 Sept. 1973	Highest Number
Wapisu Lake	414	46	414
Rat River	200	87	200
Burntwood River (upstream)	n.s.	n.s.	n.s.
Threepoint Lake	15	377	377
Burntwood River (downstream)	57	502	502
Wuskwatim Lake	72	944	944
Honeymoon Lake	n.s.	n.s.	n.s.
Notakikwaywin Lake	n.s.	n.s.	n.s.
TOTALS	**758**	**1956**	**1956**

n.s. = not surveyed

Table 7. Numbers of diving ducks observed during pre-diversion surveys of the Rat-Burntwood rivers area.

Area	7 Sept. 1973	26 Sept. 1973	Highest Number
Wapisu Lake	12	22	22
Rat River	10	0	10
Burntwood River (upstream)	n.s.	n.s.	n.s.
Threepoint Lake	20	31	31
Burntwood River (downstream)	5	0	5
Wuskwatim Lake	265	169	265
Honeymoon Lake	n.s.	n.s.	n.s.
Notakikwaywin Lake	n.s.	n.s.	n.s.
TOTALS	**312**	**222**	**312**

n.s. = not surveyed

Table 8. Numbers of ducks observed during pre-diversion surveys of the Rat-Burntwood rivers area.

Area	7 Sept. 1973	26 Sept. 1973	Highest Number
Wapisu Lake	426	95	426
Rat River	235	87	235
Burntwood River (upstream)	n.s.	n.s.	n.s.
Threepoint Lake	35	408	408
Burntwood River (downstream)	97	502	502
Wuskwatim Lake	388	1127	1127
Honeymoon Lake	n.s.	n.s.	n.s.
Notakikwaywin Lake	n.s.	n.s.	n.s.
TOTALS	**1181**	**2219**	**2219**

n.s. = not surveyed

Table 9. Numbers of Canada geese observed during pre-diversion surveys of the Rat-Burntwood rivers area.

Area	7 Sept. 1973	26 Sept. 1973	Highest Number
Wapisu Lake	0	0	0
Rat River	90	20	90
Burntwood River (upstream)	n.s.	n.s.	n.s.
Threepoint Lake	6	90	90
Burntwood River (downstream)	0	0	0
Wuskwatim Lake	12	0	12
Honeymoon Lake	n.s.	n.s.	n.s.
Notakikwaywin Lake	n.s.	n.s.	n.s.
TOTALS	108	110	110

n.s. = not surveyed

Table 10. **Changes in waterfowl numbers in the Rat-Burntwood rivers area from pre-diversion to post-diversion based on highest survey counts.**

Species by Area	Pre-diversion 1973	Post-diversion 1988	% Change
WAPISU LAKE			
Canada Geese	0	0	0.0
Dabblers	414	25	-94.0
Divers	22	2	-90.9
All ducks*	426	27	-93.7
RAT RIVER			
Canada Geese	90	8	-91.1
Dabblers	200	30	-85.0
Divers	10	14	40.0
All ducks*	235	44	-81.3
THREEPOINT LAKE			
Canada Geese	90	0	-100.0
Dabblers	377	0	-100.0
Divers	31	1	-96.8
All ducks*	408	5	-98.8
LOWER BURNTWOOD R.			
Canada Geese	0	0	0.0
Dabblers	502	25	-95.0
Divers	5	1	-80.0
All ducks*	502	26	-94.8
WUSKWATIM LAKE			
Canada Geese	12	75	525.0
Dabblers	944	1	-99.9
Divers	265	20	-92.5
All ducks*	1127	20	-98.2

* includes dabblers, divers, & unidentified ducks.

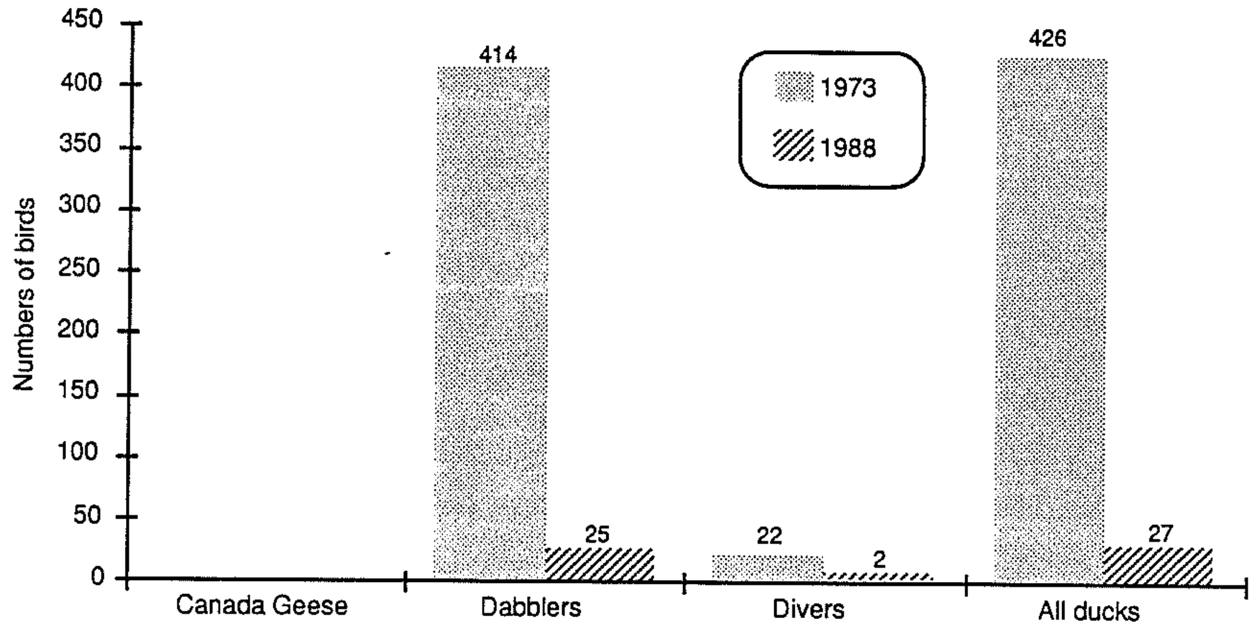


Figure 10. Pre- and post-diversion use of Wapisu Lake area by waterfowl based on highest survey counts.

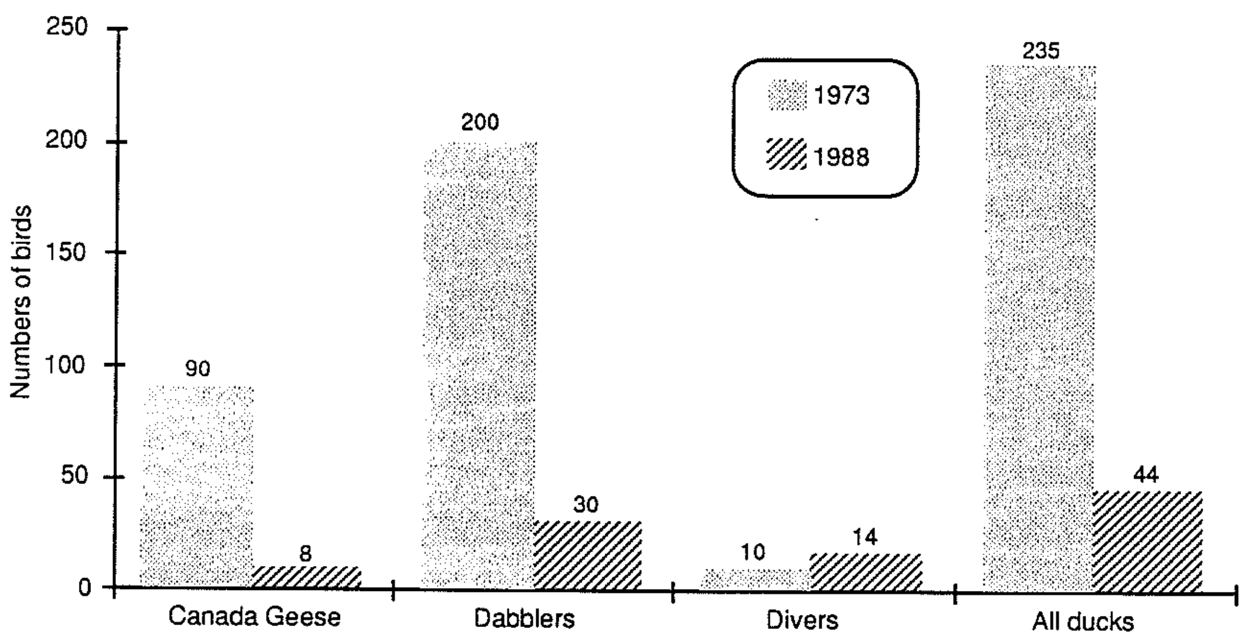


Figure 11. Pre- and post-diversion use of Rat River area by waterfowl based on highest survey counts.

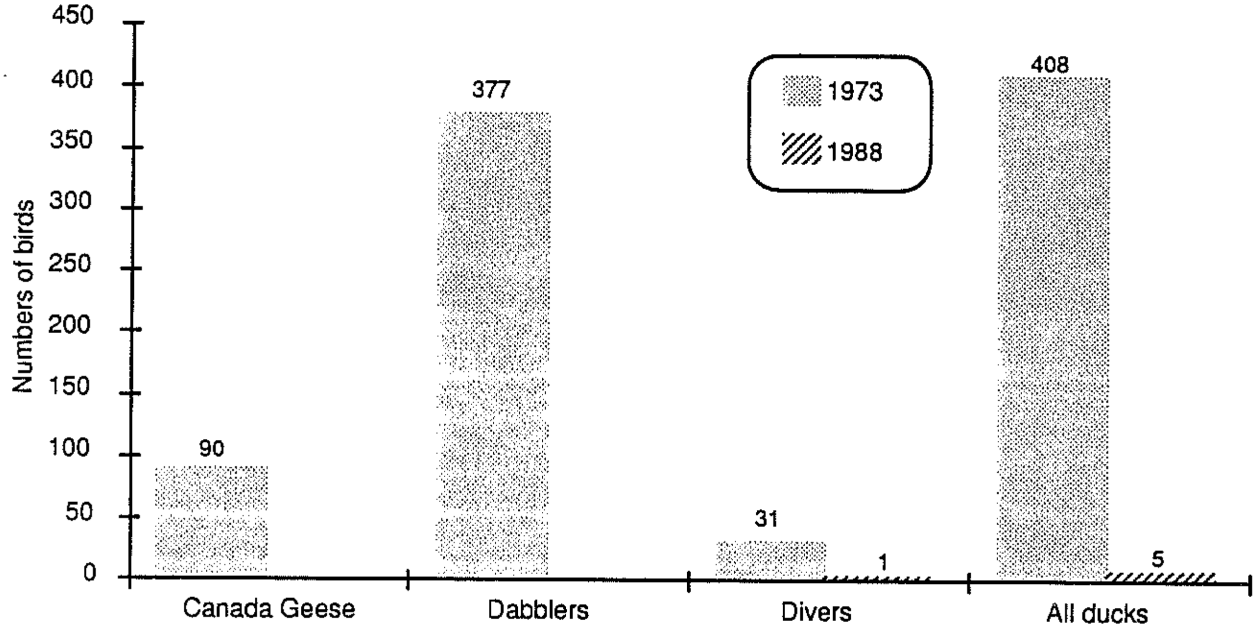


Figure 12. Pre- and post-diversion use of Threepoint Lake area by waterfowl based on highest survey counts.

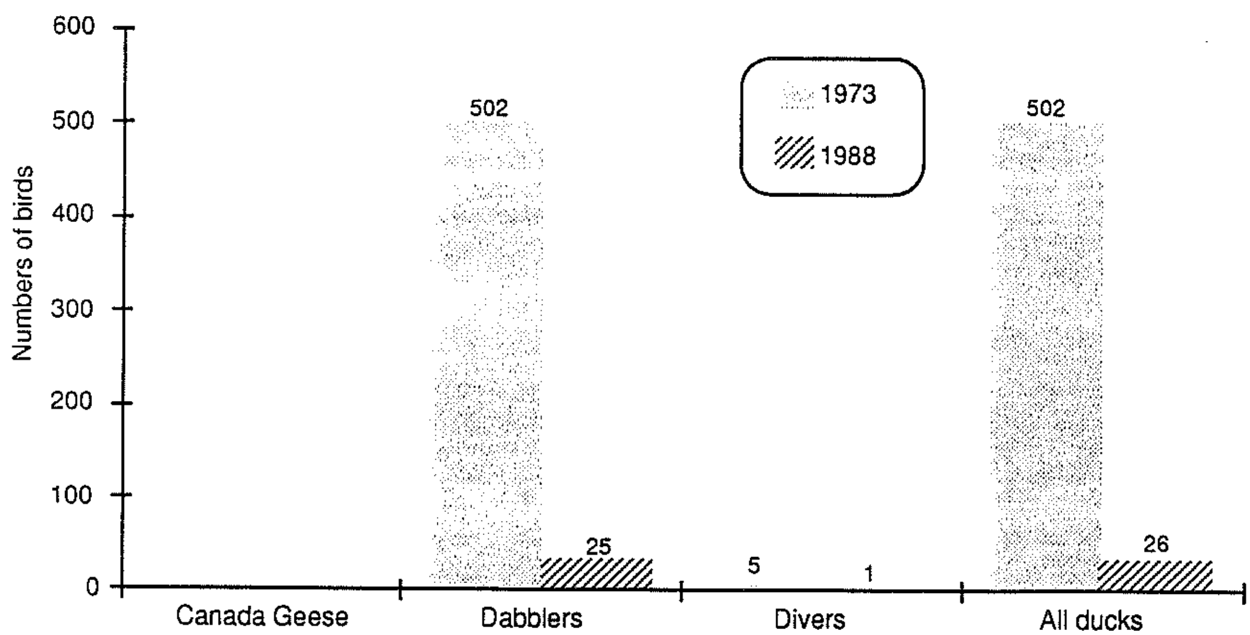


Figure 13. Pre- and post-diversion use of lower Burntwood River area by waterfowl based on highest survey counts.

system as a whole (Table 11, Figure 15), with dabblers accounting for the greatest change (-96.7 percent).

While numbers of Canada geese were 57 percent lower overall in the Rat-Burntwood areas in 1988, based on comparison of highest survey counts, one cannot reasonably assume that such change represents an actual decline in goose use, given the variability in observations between years and sites.

In general, it is risky to draw broad conclusions about changes in animal numbers from two years of survey data represented by a limited number of "point-in-time" counts. Nevertheless, in the case of dabbling ducks, the between-year differences are large enough to suggest use of the Rat-Burntwood system by dabblers has declined following diversion. The survey data also suggest that diving duck use of shoreline has declined since diversion, particularly in Wuskawatim Lake; however, as with Canada geese, the small numbers of divers recorded in both survey years make it difficult to state definitively that a decline has occurred in all survey areas.

7.2 Possible Causes of Waterfowl Use Changes

The following possible causes for the observed changes in waterfowl numbers, and especially of dabbling ducks, in the Rat-Burntwood system include:

1) Regional changes in waterfowl populations
2) Modifications to the physical environment and habitat associated with the Churchill River Diversion.

Table 11. Summary of changes in waterfowl use of the Rat-Burntwood area for lakes represented by both pre- and post-diversion counts.

Species	Pre-diversion 1973	Post-diversion^ 1988	% Change
Canada Geese	192	83	-57
Dabblers	2437	81	-97
Divers	333	38	-89
All ducks*	2698	122	-95

* includes dabblers, divers & unidentified ducks.

^ numbers of each species represent highest counts from Tables 2 through 5 and reflected in the post-diversion column of Table 10.

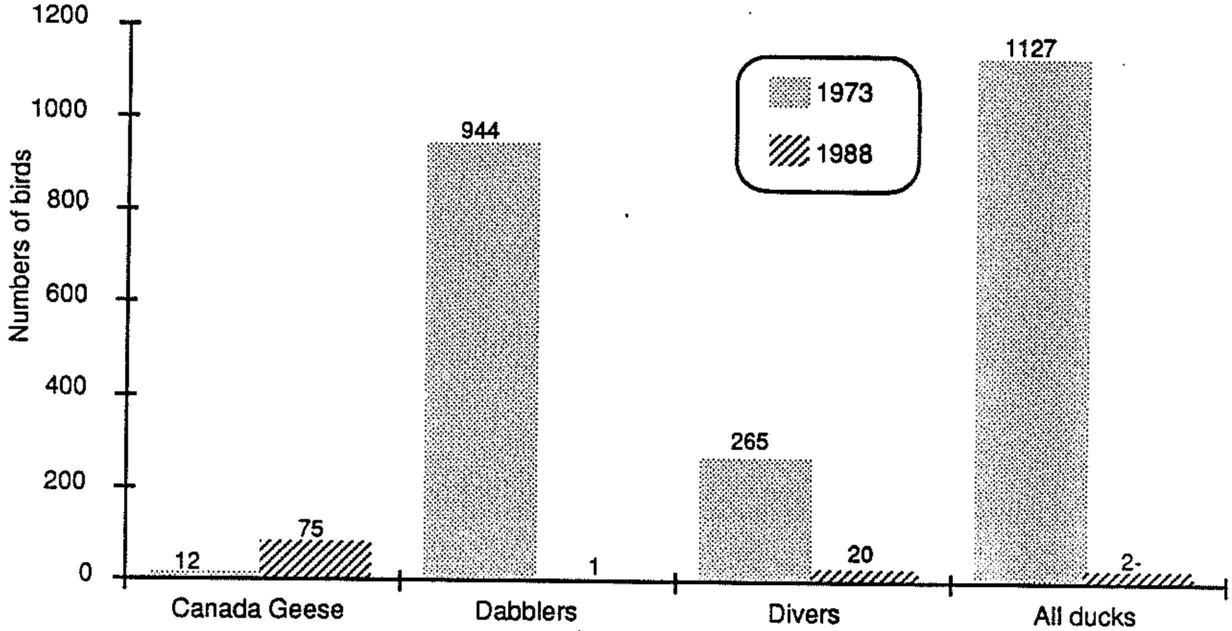


Figure 14. Pre- and post-diversion use of Wuskwatim Lake area by waterfowl based on highest survey counts.

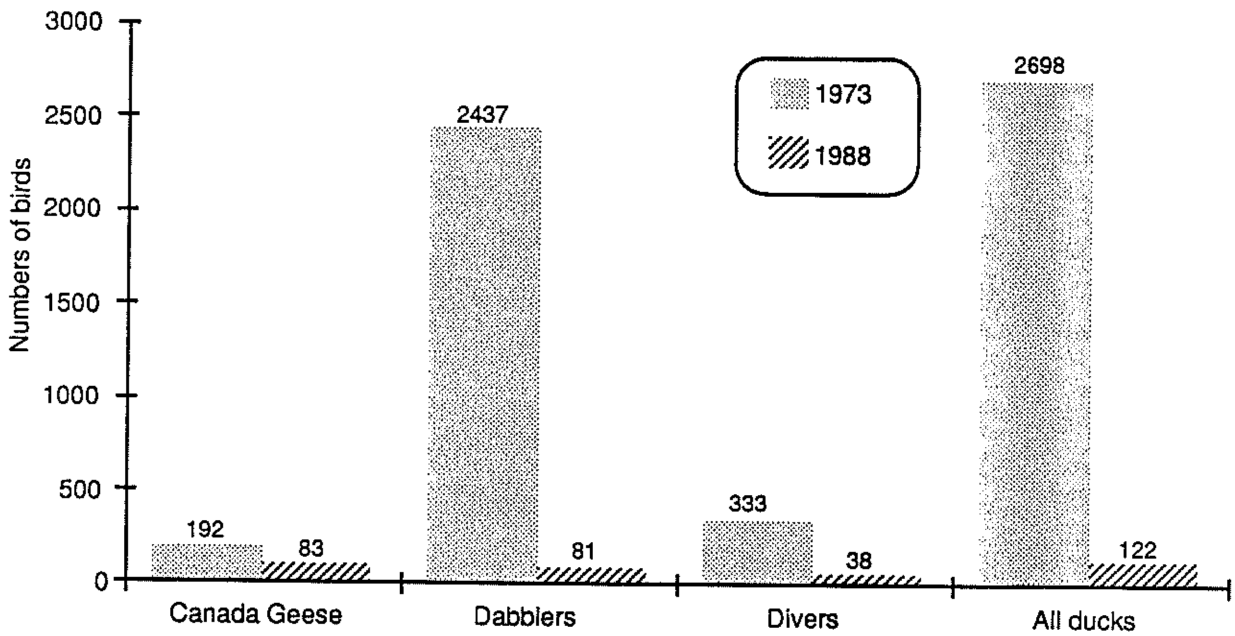


Figure 15. Pre- and post-diversion use of Rat-Burntwood area by waterfowl based on highest survey counts.

7.2.1 Regional Waterfowl Population Changes

The higher density of ducks recorded at Notakikwaywin Lake, located about 15 km from the Burntwood River system, indicates that the decline in duck use of the Rat-Burntwood rivers system following diversion is not simply a reflection of regional changes in duck populations. Nevertheless, one cannot overlook the possibility that regional declines in waterfowl populations could have occurred because of overexploitation by hunters, or as a result of a widespread reduction in the availability or quality of habitat. If waterfowl populations had declined throughout northern Manitoba, one would expect such declines to be represented in waterfowl numbers in more localized areas such as the Rat-Burntwood area.

To determine whether the observed reduction in waterfowl use of the Rat-Burntwood area could be attributed to a more widespread, regional decline in waterfowl populations, data collected annually by the U.S. Fish and Wildlife Service, during breeding pair surveys in Stratum 24, northern Manitoba (Figure 16), were examined. These data yield adjusted population estimates for ducks and indicated breeding pair numbers for Canada geese for the stratum. It must be stressed that these data estimate the relative size of breeding populations of ducks and geese and cannot be used to represent numbers or distribution of migrating or staging waterfowl. For the purposes of this study, the data are useful as broad indicators of regional population size and trends.

Data for the period 1964-75 were selected to describe regional population levels prior to the CRD project, while data for 1977-88 represent regional population levels following diversion. The diversion was phased in over 1976 and 1977 (J. Funnell, pers. comm.); data for those years were not included in the analysis. Duck and Canada goose population estimates for these two periods are given in Table 12. Regional population changes in ducks and Canada geese that have occurred from 1955 to 1988 are displayed in Figures 17 and 18.

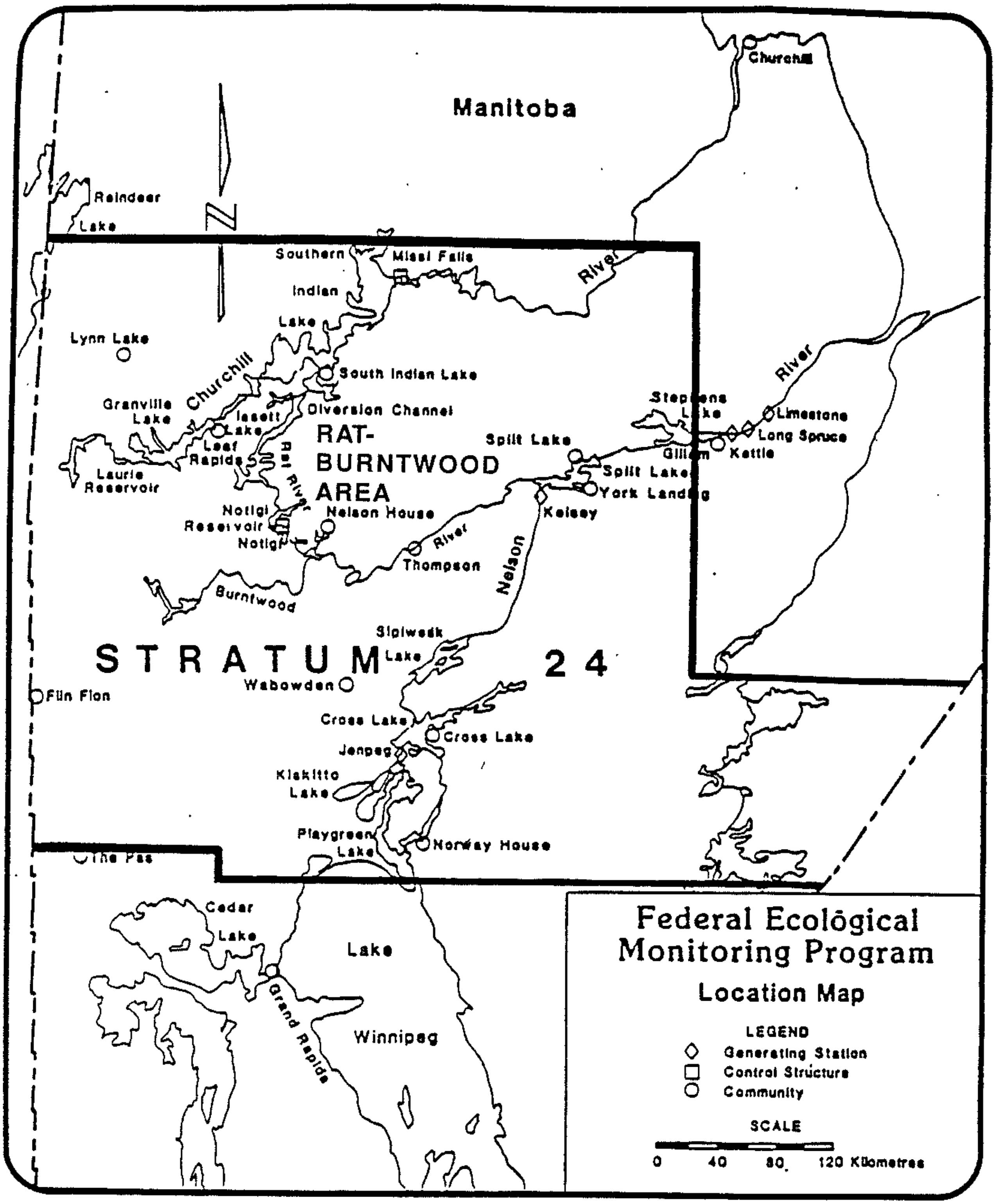


Figure 16. Location of waterfowl survey stratum 24.

Table 12. Adjusted population estimates (thousands) for dabblers, divers, and Canada geese in stratum 24 during pre-diversion (1964-75) and post-diversion (1977-88) periods (from U.S. Fish & Wildlife Service).

PRE-DIVERSION	1964	1965	1966	1967	1968	1969	1970	1971	1972	1973	1974	1975	MEAN
dabblers	228.9	404.4	205.0	716.0	1122.9	1102.3	1458.7	631.2	440.2	500.3	345.1	358.5	626.1
divers	198.2	276.7	228.3	233.0	587.7	653.3	788.0	447.4	513.8	375.5	270.2	383.2	412.9
Totals	**427.1**	**681.1**	**433.3**	**949.0**	**1710.6**	**1755.6**	**2246.7**	**1078.6**	**954.0**	**875.8**	**615.3**	**741.7**	**1039.1**
Canada geese	4560	6952	5145	5050	9504	15606	5925	4700	2498	1845	1590	2041	5451.3

POST-DIVERSION	1977	1978	1979	1980	1981	1982	1983	1983	1985	1986	1987	1988	MEAN
dabblers	708.9	1101.5	935.5	799.8	837.2	561.6	584.4	347.2	372.6	248.6	441.3	588.6	627.3
divers	536.2	376.2	336.9	387.6	746.3	329.4	531.7	319.4	359.1	534.1	339.8	288.2	423.7
Totals	**1245.1**	**1477.7**	**1272.4**	**1187.4**	**1583.5**	**891.0**	**1116.1**	**666.6**	**731.7**	**782.7**	**781.1**	**876.8**	**1051.0**
Canada geese	4444	6750	9743	5125	5681	2840	5000	2525	5517	1914	5345	8104	5249.0

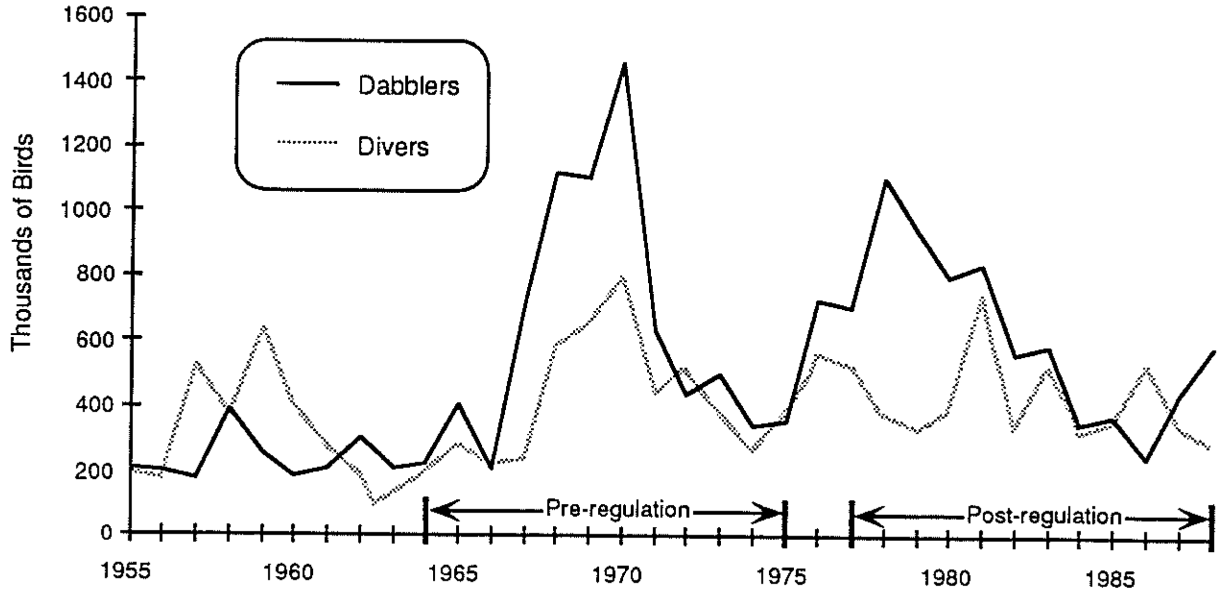


Figure 17. Adjusted population estimates of dabblers and divers in stratum 24 (northern Manitoba), 1955-1988.

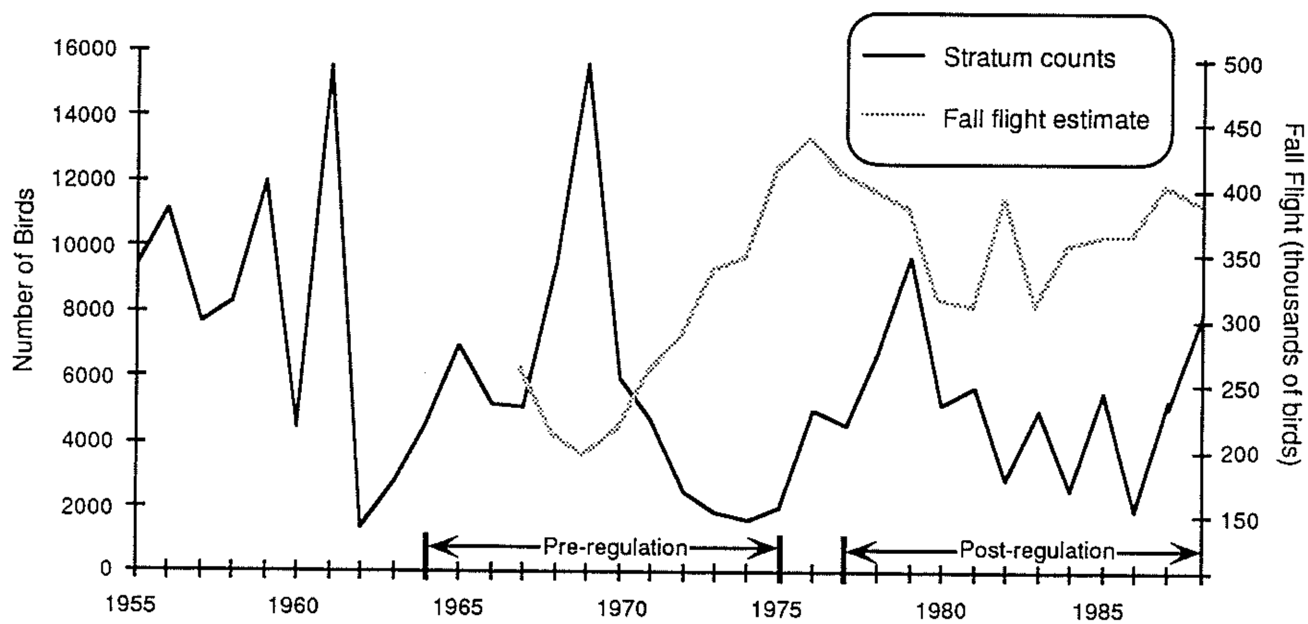


Figure 18. Estimates of indicated Canada goose pairs in stratum 24 (northern Manitoba), 1955-1988, and the estimated fall flight of Canada geese, Eastern Prairie Population, 1967-1988, based on production survey data.

Figure 17 shows that large year-to-year fluctuations in regional dabbler and diver populations occurred during both the pre- and post-diversion periods. However, when the 12-year means for the two periods are compared, only a small change in numbers is evident from the pre-diversion period to the post-diversion period for both dabblers and divers (Table 13). Results of the 1973 and 1988 surveys for the Burntwood area included for comparison. Overall, the dabbler population increased by 0.2 percent across Stratum 24 while the diver population declined by 2.6 percent.

One must keep in mind, however, that the means on which these changes are based will vary somewhat as individual year's counts are added or subtracted. The main point to be made is that numbers of breeding waterfowl in the study region have neither declined nor increased over the long term, according to Stratum counts.

Figure 18 shows that, as with regional dabbler and diver populations, large year-to-year fluctuations in the Canada goose breeding population have occurred in stratum 24 during both the pre-diversion and post-diversion periods, but do not indicate any long-term trends toward population increase or decline.

For comparison, fall-flight estimates for the Eastern Prairie Population (EPP) of Canada Geese based on production surveys carried over an area that broadly overlaps Stratum 24 (EPP Progress Report, 1988) have also been included in Figure 18. Disregarding the obvious differences between the two sets of data, there is no indication of a downward trend in the EPP fall-flight forecasts, which have remained relatively stable -- with moderate between-year fluctuations -- since 1975.

In conclusion, there is no indication that the observed reduction in waterfowl use in the Rat-Burntwood area reflects a long-term decline in regional populations as represented by the Stratum 24 counts, since no such trends are evident in the stratum data. At the same time, one must be cautious about interpreting the significance of the 97 percent change in dabbler numbers derived from 1973 and 1988 surveys, because large between-year changes are not uncommon in waterfowl counts derived from aerial surveys.

Table 13. Changes in regional (Stratum 24) and local (Rat-Burntwood) waterfowl use from pre-diversion (1973) to post-diversion (1988).

Waterfowl	Stratum 24: 12-year Means* 1965-75 (pre)	1978-88 (post)	% Change	Rat-Burntwood Counts** 1973 (pre)	1988 (post)	% Change
Dabblers	626.1	627.3	0.2	2437	81	-96.7
Divers	412.9	423.7	2.6	333	38	-88.6

* adjusted population estimates (thousands)
** numbers represent birds/km

For example, comparison of any pre-diversion year with any post-diversion year estimate using the Stratum 24 data can yield dramatically different results that may or may not have anything to do with changes in regional water conditions: comparison 1986 and 1970 Stratum counts suggest a 78 percent reduction in dabbling ducks, whereas comparison of 1978 and 1966 counts indicate a 437 percent increase. In the absence of counts compiled over several years, one would be hard-pressed to draw meaningful conclusions from these point-in-time data.

7.2.2 Modifications to the Habitat

Extensive modifications to terrestrial and aquatic habitat have occurred due to increases in water levels that have accompanied the diversion. The magnitude of these increases is illustrated by the water level data collected at Water Survey of Canada's Footprint Lake at Nelson House hydrometric station (# 05TF001). The historic record from 1969 to 1988 was divided into pre-diversion (1969-75) and post diversion (1978-88) periods. Data for 1976 and 1977 were excluded because they were considered to be transitional between pre- and post-diversion conditions. Connection of Southern Indian Lake to Rat River occurred on June 2, 1976, and the diversion was phased in over 1976 and 1977 (J. Funnell, pers. comm.).

Mean monthly water levels at Footprint Lake during the pre- and post-diversion periods are presented in Table 14 and Figure 19. These data show that, when the historic record is considered, mean monthly post-diversion water levels were between 3.81 and 5.45 m higher than mean monthly pre-diversion water levels. In August and September, when waterfowl have traditionally used the Rat-Burntwood area for staging prior to migration, mean monthly post-diversion water levels were approximately 4.5 m higher than the mean levels before the diversion. (Because gaps exist in the daily record of water levels at Footprint Lake, the above means must be considered approximate).

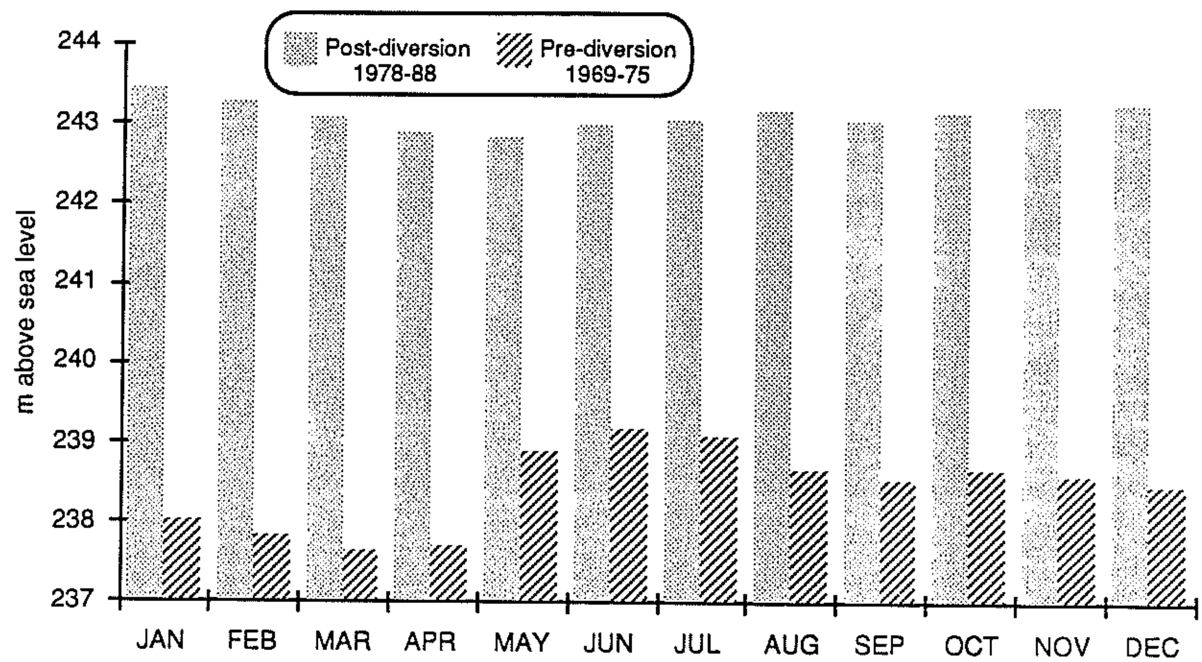
These comparisons indicate that, following diversion, water levels along the diversion route rose considerably. The effect of this increase in water levels on the environment has been experienced by members of the Nelson House Band and was described during the hunter interviews (see section 6.2 of this report). The changes to shoreline habitat resulting from the rise in water levels were also noted during the aerial waterfowl surveys. Well-vegetated marsh areas that used to exist in shallow bays of lakes, such as Wapisu Lake, and along the Rat and Burntwood rivers, have been inundated. In contrast, abundant marsh habitat was observed in shallow water areas of Notakikwaywin Lake which remains unaffected by the diversion. Pondweeds and other submerged vegetation were clearly visible at Notakikwaywin Lake but were not observed anywhere in the Rat-Burntwood area.

Table 14. Pre- and post-diversion water levels (m asl) at Footprint Lake-Nelson House hydrometric station (#05TF001).

	JAN	FEB	MAR	APR	MAY	JUN	JUL	AUG	SEP	OCT	NOV	DEC
A. HISTORIC												
1978-88 (Post)	243.42	243.26	243.08	242.89	242.81	242.98	243.05	243.18	243.05	243.12	243.22	243.27
1969-75 (Pre)	238.02	237.82	237.63	237.70	238.89	239.17	239.07	238.65	238.51	238.66	238.57	238.45
Difference	5.40	5.44	5.45	5.19	3.92	3.81	3.98	4.53	4.54	4.46	4.65	4.82

* Due to gaps in monthly water level records, monthly means could not be calculated for every year; therefore the multi-year means shown here were derived from discontinuous data and must be considered approximate.

BASED ON HISTORIC RECORD



BASED ON SELECTED YEARS

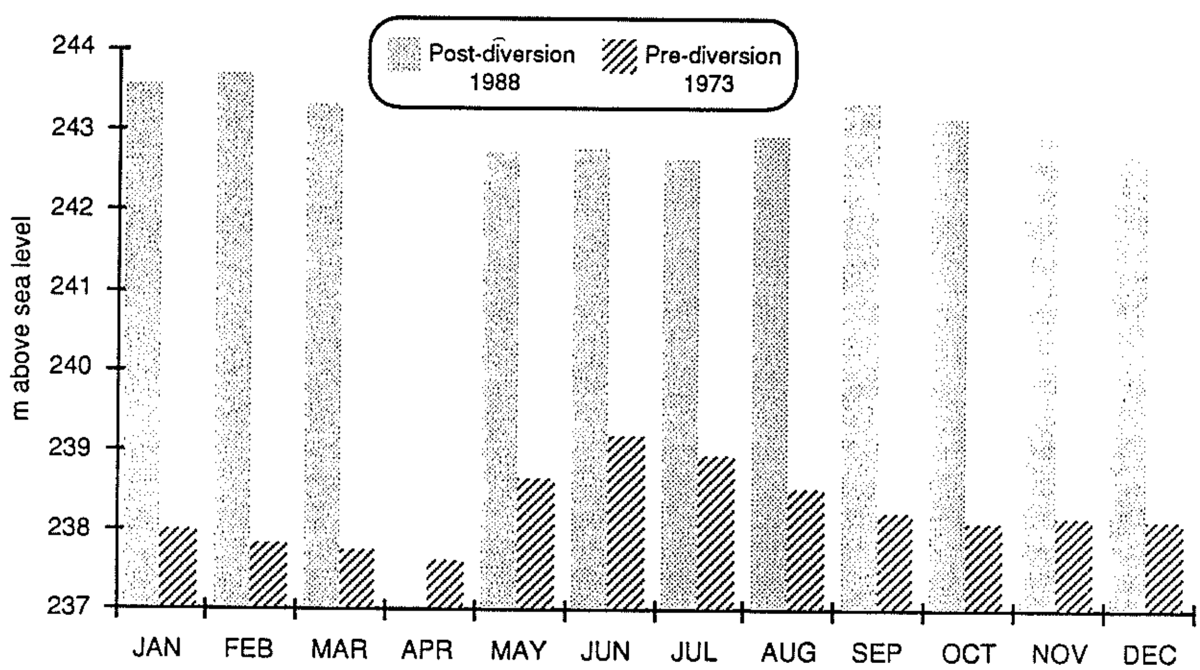


Figure 19. Pre- and post diversion water levels at Footprint Lake (Nelson House) hydrometric station (#05TF001).

While former shallow water areas have been transformed into deep water areas, a large proportion of new shorelines associated with shallow water are ill-defined and are characterized by dead trees (fallen or standing), tree stumps and other debris -- habitat not conducive to attracting waterfowl.

In their assessment of shoreline conditions on Footprint and Honeymoon lakes, Kellerhals Engineering (1988) noted that before diversion, areas with shallow water depths or floating or submerged vegetation were limited to low-gradient deltas near mouths of streams, with cleared or eroded shorelines confined to the Nelson House area. After diversion, the length of shoreline characterized by shallow water or floating/submerged vegetation increased from 14 km (pre-diversion) to 83 km. The shoreline length with standing debris rose from 0.2 to 56 km.

From their analysis, Kellerhals Engineering (1988) concluded:

> *that the Churchill-Nelson Diversion has had a widespread impact on shoreline characteristics. The extensive areas of newly eroding shoreline will act as a continuing source of sediment until such time as they restabilize. Areas of shallow water, floating or submerged vegetation and inundated debris restrict access to the shore zone and dead-heads will interfere with open-water travel for years to come.*

It is important to remember, however, that, in the long term, increases in the shallow-water zone associated with higher water levels can be expected to provide at least as much habitat suitable for waterfowl (including diving ducks) as existed before regulation, although the time required for shorelines to stabilize will span several decades.

The key habitat attributes that formerly attracted ducks prior to fall migration are now absent. The dramatic reduction in use of the area by waterfowl would appear to support the idea that habitat appearance is a major factor influencing waterfowl use.

The influence of landscape and habitat characteristics on the attractiveness of an area to waterfowl and other bird species was discussed by I.D. Systems (1991) in relation to the effects of the Lake Winnipeg Regulation project on waterfowl use of the Outlet Lakes area. Some of this discussion is germane to understanding the effect of habitat modifications, associated with the CRD, on waterfowl use of the Rat-Burntwood area and is repeated here.

The mechanics of bird navigation and habitat selection is poorly understood despite the fact that many scientific papers have been published on these subjects (Bellrose 1976). Hilden (1965) provides some useful insights in his review of habitat selection mechanisms in birds during the breeding season. It is reasonable to assume that similar mechanisms operate with respect to selection of staging habitat by waterfowl during fall migration.

Hilden refers to "proximate" and "ultimate" factors that together determine the selection of suitable habitat. Ultimate factors such as food and shelter are essential to the survival of a species. Proximate factors refer to characteristic species-specific stimuli provided by habitat that release the "settling reaction". These factors include: landscape, terrain, feeding and drinking sites, and other animals.

Not every suitable habitat need possess all the features characteristic of the optimal environment. Rather, the positive and negative habitat characteristics are summed and, when the combined effect of the stimuli exceeds the threshold of the settling reaction, the habitat is selected. Similarly, one key stimulus, such as adverse landscape appearance, may out-weigh the other stimuli; in that situation, the combined effect of other positive stimuli would not be sufficient to induce the bird to "settle" in that particular habitat. Studies on wild geese conducted by Barry (1962) and Wood (1964) concluded that, where certain proximate factors such as appropriate nesting habitat conditions are lacking, ovulation can be inhibited and the nesting cycle terminated. Marshall (1952) speculated that inhibitions to breeding in birds appear to be brought about by the lack of one or more important environmental stimuli necessary to complete the reproductive cycle.

Habitat selection is described by Hilden as being a two-stage process. The first stage, consisting of settling down and exploring a particular habitat, is released by features of the landscape and general characteristics of the terrain. Whether the area in question is chosen or rejected depends on how closely certain of its details conform to other proximate factors in the bird's habitat selection mechanism.

The Rat-Burntwood area provides numerous environmental stimuli related to habitat quality and appearance that would likely cause ducks and geese to seek other more suitable areas. These negative stimuli include: the absence of submerged vegetation; the absence of emergent aquatic vegetation; the presence of inundated trees and debris along the shoreline; and the presence of eroding and slumping banks along the shoreline. The only positive environmental stimulus from the standpoint of waterfowl is the presence of water, which in the Rat-Burntwood area apparently is not sufficient to elicit a "settling reaction" in fall staging waterfowl.

8.0 SUMMARY AND CONCLUSIONS

8.1 Project Impacts on Waterfowl

Following completion of the Churchill River Diversion project in 1977, waterfowl use of the Rat-Burntwood rivers system in the fall has declined. Numbers of ducks observed along shorelines of lakes and rivers in the Rat-Burntwood area in the fall of 1988 were approximately 95 percent lower than the numbers recorded prior to the CRD project in the fall of 1973. Dabbling ducks have experienced the greatest reduction - almost 97 percent - while diving ducks declined by about 89 percent. Canada goose use of the area dropped by 57 percent, although this change is of questionable significance, given the limited number and variable distribution of birds recorded during 1973 and 1988 surveys.

The observed decline in the use of the Rat-Burntwood area by dabblers in particular cannot be attributed to regional waterfowl populations documented in northern Manitoba. There is no indication that regional duck populations in the study region have increased or declined in the long term, although their numbers have fluctuated dramatically from year to year.

Analysis of water level data collected at Footprint Lake indicates that, following the diversion, water levels along the Rat-Burntwood system have risen 3.5 to 5.8 m. The elevated water levels have flooded natural shorelines and created poor quality habitat characterized by dead trees, fallen timber and other debris. As a result, habitat modifications resulting from the Churchill River Diversion may have rendered the Rat-Burntwood less attractive for staging dabblers and (to a lesser extent) divers.

8.2 Project Impacts on the Nelson House Band

Nelson House residents interviewed during the study indicated that presence of fallen trees and other debris has made boat travel and access to traditional waterfowl harvesting areas difficult or impossible. The reduction in waterfowl availability, combined with boat navigation and access problems, have limited opportunities for Nelson House Band members to hunt and view waterfowl.

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APPENDIX A

NORTHERN FLOOD AGREEMENT

Background

The **Northern Flood Agreement (NFA)** was signed on December 16, 1977 by Canada, Manitoba, Manitoba Hydro and the Northern Flood Committee to establish a mechanism for compensating members of the Nelson House, Norway House, Cross Lake, Split Lake and York Landing (formerly York Factory) bands for adverse effects caused by the hydroelectric development.

The Preamble of the Agreement lays out the premises and principles which together provide a basis for the Agreement's provisions. Clauses C, D, E and G of the Preamble are particularly relevant to the present study and are reproduced below verbatim:

> *C.* *As a result of the modification of the water regime, adverse effects have occurred and may continue to occur, on the lands, pursuits, activities and lifestyles, of the residents, individually and collectively, of the Reserves of Cross Lake, Nelson House, Norway House, Split Lake and York Landing ("the Reserves" as hereinafter defined);*
>
> *D.* *The parties wish to ensure that all persons as defined herein, who may be, or have been, directly or indirectly, adversely affected by the* [Lake Winnipeg Regulation and Churchill River Diversion] *Project shall be dealt with fairly and equitably;*
>
> *E.* *Uncertainty as to the effects of the Project, with respect not only to the Project as it exists at the date of this Agreement but also as it may develop in he future, is such that it is not possible to foresee all the adverse results of the Project nor to determine all those persons who may be affected by it, and, therefore it is desirable to establish through the offices of a single arbitrator a continuing arbitration instrument, to which any person may submit a claim,*

and as well as to fully empower such arbitrator to fashion a just and appropriate remedy;

G. *Canada, by virtue of its jurisdiction and responsibility for Indians and land reserved for Indians, is committed to playing an active role in providing opportunity for the continued viability of the communities and, in particular but without limitation, in making available resources and expertise to the communities in planning and improving the social and economic conditions of the communities, and in ensuring that the special rights of Indians, including those arising from Treaty 5, are adequately protected.*

Contractual Obligations of the Northern Flood Agreement

The provisions of the **NFA** are arranged in a set of 23 Articles which represent contractual obligations. Three subarticles ultimately lead to the initiation of the present study and are, therefore, reproduced below verbatim:

17.1 *Hydro and Canada and Manitoba, severally and jointly, undertake to implement such recommendations of the Lake Winnipeg, Churchill and Nelson Rivers Study Board Report which affect the communities and which fall within their respective or joint jurisdictions.*

17.5 *In particular but without limitation, monitoring of adverse effects of the Project pursuant to the Lake Winnipeg, Churchill and Nelson Rivers Study Board recommendations shall be planned and implemented so as to provide information as may be necessary to give effect to this Agreement.*

18.2 *Canada and Manitoba recognize that the Project is intended to benefit all citizens of Canada, and most particularly of Manitoba, on the one hand, and that the resource users have been and may continue to be adversely affected on the other hand, and that it is in the public interest to ensure that any damage to the interests, opportunities, lifestyles and assets of those adversely affected be compensated appropriately and justly.*

Claim 18

In December 1981, the five **NFA** Indian Bands and the Northern Flood Committee filed a claim (Claim 18) alleging that Canada, Manitoba and Manitoba Hydro (the Respondents) had failed to implement recommendations of the Lake Winnipeg, Churchill and Nelson Rivers Study Board and to report to the **NFA** bands on implementation measures.

In particular, they claimed that the Respondents had "failed to establish a mechanism to deal with monitoring and analysis of ongoing social and economic changes related to hydro-electric development and more generally, northern development pursuant to Recommendation 5(c) of the Study Board". In addition, they claimed that the Respondents had "failed to develop and implement a long-term coordinated ecological monitoring and research program to allow impact evaluation and to assist in the ongoing management of the area affected by the Hydro Project pursuant to Recommendation 10 of the Study Board".

Failure to implement these two recommendations meant that the adverse ecological effects of the Project, and the associated adverse social and economic effects, had not been monitored and the information necessary to give effect to the **NFA** not provided. Consequently, the requirements of **NFA** sub-article 17.5 had not been met. Furthermore, without this information, it has not been possible to determine to what extent damages to the interests, opportunities, lifestyles and assets of resource users have occurred and, therefore, the level of compensation that is necessary (sub-article 18.2).

Filing of the claim resulted ultimately in approval of funds by Treasury Board on February 6, 1986, for implementation of the **NFA** over the period 1986-91 (T.B. Minute 800953). Availability of these funds permitted the Canadian Wildlife Service to initiate studies of the adverse effects of the project on waterfowl use of the Rat-Burntwood rivers system in the fall of 1988.

Impact Management

Impact management refers to a mechanism, or set of mechanisms, by which the adverse impacts of the CRD project may be managed, whether they be biophysical impacts (e.g., adverse effects on waterfowl) or socio-economic-cultural impacts (e.g., reduced waterfowl hunting opportunities). Suitable and adequate compensation for adverse effects on **NFA** band would constitute redress for damages suffered by the bands.

NFA sub-article 24.8 refers to three forms of compensation or redress for damages suffered:

> *24.8 Because mitigatory and/or remedial measures are more likely to have a lasting beneficial effect on the viability of a community and/or on individual residents than monetary compensation, such measures shall be preferred and only where mitigatory and/or remedial measures are not feasible or fail in effectiveness shall monetary compensation be ordered in lieu thereof in respect of any adverse effect.*

Article 1 of the **NFA** provides definitions for the terms "mitigatory measure" and "remedial measure":

> *1.9 "Mitigatory measure" means any work, program or measure which is designed or intended to diminish, prevent, or ameliorate any adverse effect of the Project.*
>
> *1.14 "Remedial measure" means any work, program or measure which is designed or intended to enhance, preserve, restore or replace in kind, wholly or in part, any property, land, land use interest or activity of any person, which has been or may be adversely affected by the Project.*

The third form of compensation is "monetary settlement" and would consist of a cash settlement for damages suffered.

NFA sub-article 24.6 provides the Arbitrator with "broad authority and power to make [compensation] awards capable of implementation and to fashion an appropriate and just remedy of any and all adverse effects of the Project on any person". Compensation awards are made on the basis of claims submitted to the Arbitrator for settlement. It should be noted that Claim 18 was not advanced for the purpose of obtaining a suitable remedy for the adverse effects experience. Rather, it was advanced to give effect to sub-article 17.5 (information) and sub-article 18.2 (compensation). The purpose of the claim was to cause the necessary <u>information</u> to be collected and made available which would allow adverse effects on **NFA** bands, and therefore damages, to be determined. This information would then be used to prepare and advance separate claims for compensation of damages.

EFFECTS OF THE LAKE WINNIPEG REGULATION PROJECT ON WATERFOWL USE OF THE OUTLET LAKES AREA AND ON NORWAY HOUSE BAND

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ABSTRACT

Surveys carried out in September 1986 and October 1987 indicated that use of the Outlet Lakes area by diving ducks (Aythyinae) in fall declined following completion of the Lake Winnipeg Regulation project. Diving duck numbers in 1986/87 were 96 percent lower than recorded during 1972/73 surveys.

The decline in diving duck use of the Outlet Lakes area cannot be attributed to a regional reduction in waterfowl populations, since no such decline was evident in regional survey data (stratum 24) compiled over the past thirty years. Analysis of water level data for Playgreen Lake indicates that slight year-round increases in water levels following Lake Winnipeg regulation have altered the quality of habitat and may have reduced the attractiveness of affected lakes to diving ducks in the fall.

Residents of Norway House interviewed during the study indicated that low diving duck numbers and high water levels have reduced opportunities for Band members to hunt and view waterfowl.

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1. INTRODUCTION

In the early 1970's, Manitoba Hydro began to proceed with developments to harness the hydroelectric potential of the Nelson River. This involved the diversion of a major portion of the lower Churchill River into the Nelson River, referred to as the Churchill River Diversion (CRD) Project, and regulation of the outflow from Lake Winnipeg through the Lake Winnipeg Regulation (LWR) Project. The LWR project is described in detail in **3. LAKE WINNIPEG REGULATION PROJECT.**

In response to concerns expressed about the effects of this development scheme on the natural environment and on northern communities that depended on the natural resources, the governments of Canada and Manitoba initiated the Lake Winnipeg, Churchill and Nelson Rivers Study. In April 1975, a report was issued by the Lake Winnipeg, Churchill and Nelson Rivers Study Board (Study Board or LWCNRSB) which described the environmental and social impacts that were expected to occur as a result of the hydroelectric development. The report also recommended measures to minimize or offset impacts and specified additional studies that should be carried out. Study Board findings and recommendations relevant to the present study are discussed in **4. PREVIOUS ENVIRONMENTAL ASSESSMENT STUDIES.**

To ensure that all members of Indian communities affected by the hydroelectric development would be adequately compensated, the **Northern Flood Agreement** (NFA) was entered into on December 16, 1977, by Canada, Manitoba, Manitoba Hydro and the Northern Flood Committee. The Northern Flood Committee was established to represent the interests of Nelson House, Norway House, Cross Lake, Split Lake and York Factory Indian Bands. Included in the **NFA** is the requirement that the effects of the development be monitored to determine how Indian Band members have been affected and how they should be compensated.

The **NFA** also established an arbitration procedure whereby injured parties could file claims and obtain suitable remedies for damages suffered. One such claim, Claim 18, led to the initiation of a Federal Ecological Monitoring Program (FEMP) in the spring of 1986. FEMP was established to address federal responsibilities under the **NFA** with respect to environmental research and monitoring. Claim 18 and the various provisions of the **NFA** are described further in **APPENDIX A: NORTHERN FLOOD AGREEMENT**.

Funds provided through FEMP enabled the present study to be initiated in response to concerns expressed by Norway House Band that opportunities for hunting waterfowl in the fall had declined following completion of the LWR project.

2. STUDY AREA

For the purposes of this study, the Outlet Lakes area consists of Playgreen, Kiskittogisu and Kiskitto lakes, bounded by Lake Winnipeg to the south and the Jenpeg Generating Station to the north (Figure 1). Cross Lake was excluded because it falls outside the study area. The Outlet Lakes receive water from the Red, Winnipeg, Saskatchewan and Dauphin river drainage basins via Lake Winnipeg.

The topography of the area reflects the underlying bedrock and the area's geomorphic history, particularly that of glacial and post-glacial periods (LWCN-RSB 1975a). The distribution of surface material has been influenced by glaciation and glacial Lake Agassiz. Organic deposits, glacial till, lacustrine clays and glaciofluvial sands comprise the surficial material. Organic deposits are widespread and consist mainly of peat. The three study lakes feature irregular bedrock shorelines that form promontories, and erodible organic and granular shorelines forming continuous sweeping arcs (LWCNRSB 1975b).

Black spruce (*Picea mariana)* and tamarack (*Larix laricina*) dominate most forest communities. Jack pine (*Pinus banksiana*), trembling aspen (*Populus tremuloides*), white spruce (*Picea glauca*), balsam poplar (*Populas balsamifera*), balsam fir (*Abies balsamea*), white birch (*Betula papyrifera*) and alder (*Alnus* spp.) occur less frequently. Willow (*Salix* spp.), alder and sedge (*Carex* spp.) communities inhabit wet or saturated sites (LWCNRSB 1975b). Marsh vegetation before regulation was characterized by a band of sedge along shorelines with an outer band of bulrush (*Scirpus* spp.) in deeper water. Floating aquatics, such as duckweed (*Lemna* spp.), were common in shallow sheltered areas; pondweed (*Potomogeton* spp.) often occurred in deeper water beyond the bulrush margin.

The climate is continental and is characterized by short, cool summers and long, cold winters. Annual precipitation averages 45 cm, 60 percent of which falls as rain between April and October (LWCNRSB 1975b).

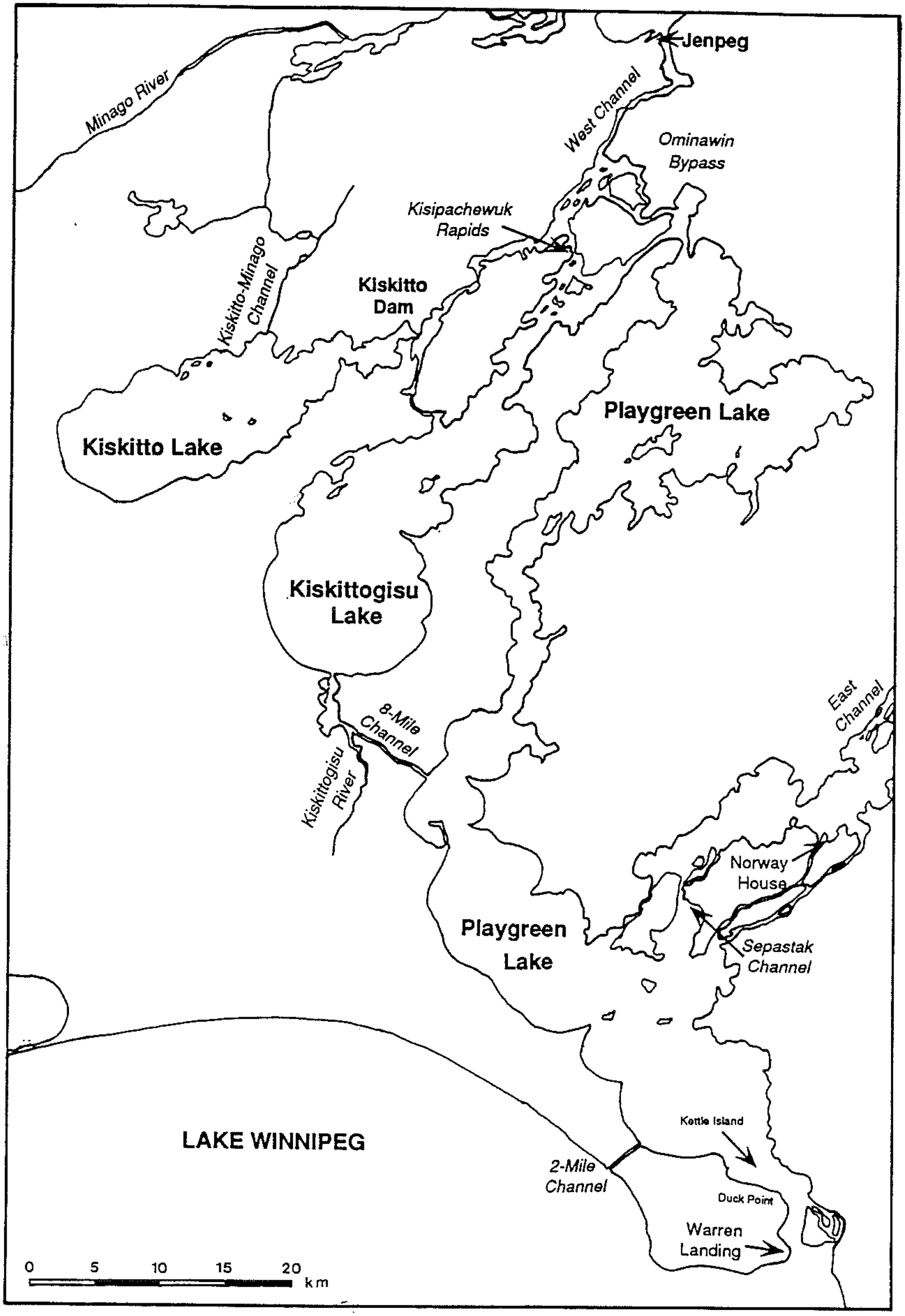

Figure 1. Outlet Lakes study area.

3. LAKE WINNIPEG REGULATION PROJECT

The following details of the LWR project were extracted from the Summary Report of the Lake Winnipeg, Churchill and Nelson Rivers Study (LWCNRSB 1975c), supplemented by other information from Manitoba Hydro.

Regulation of the outflow of Lake Winnipeg is a major component of hydroelectric development on the Nelson River. Before regulation, natural seasonal variations in outflow of Lake Winnipeg were out of phase with power demand. The regulation project was designed to ensure that releases from Lake Winnipeg could be increased during the winter, the higher power demand period, and that water could be stored from one year to the next in the event of low runoff conditions in the drainage basin. To accomplish this, it was necessary to increase the discharge capacity from the Lake and to install a structure across the outlet that would permit flow releases from the Lake to be controlled.

The Jenpeg Generating Station was constructed on the Nelson River West Channel and controls approximately 30 percent of the outflow from Lake Winnipeg (Figure 1). The remaining flow passes down the uncontrolled East Channel through Pipestone Lake into Cross Lake. In addition to controlling flows, Jenpeg can also produce up to 168 mW of power.

Three new channels were excavated to increase the capacity to discharge water from Lake Winnipeg to the Nelson River: 2-Mile Channel, 8-Mile Channel and the Ominawin Bypass (Figure 1). In addition, limited channel improvement work was carried out at Kisipachewuk Rapids. The 2-Mile Channel increased the discharge capacity between Lake Winnipeg and Playgreen Lake, while 8-Mile Channel increased the discharge capacity between Playgreen and Kiskittogisu lakes. The Ominawin Bypass diverts a portion of the flow around Ominawin Rapids and into the West Channel.

The Kiskitto Lake dam, and associated dyke system, is an integral part of the West Channel control system which prevents flows from bypassing the Jenpeg Generating Station (Figure 1). The dam and dykes isolate Kiskitto Lake from the raised water levels in the West Channel. A gated conduit installed in the dam permits flows from the West Channel into Kiskitto Lake to augment natural inflow into the lake, when required. Outflow from the lake is discharged through a stop-log control structure into the Kiskitto-Minago Channel. This channel was constructed to connect Kiskitto Lake to Black Duck Creek and allows Kiskitto Lake outflows to enter the Minago River and Cross Lake.

Implementation of the LWR project began in May 1972 with construction of the Jenpeg tailrace cofferdam. By July 1976, all facilities were sufficiently in place to permit full, unrestricted regulation of Lake Winnipeg for the first time.

4. PRE-PROJECT ENVIRONMENTAL ASSESSMENT

4.1 Lake Winnipeg, Churchill and Nelson Rivers Study

4.1.1 Study Findings

In its investigations of the physical impacts of Lake Winnipeg regulation, LWCNRSB (1975b) predicted that changed water levels would affect marsh zones in the north and south basins of Playgreen Lake. Altered water level regimes were expected to "not only reduce the total width of the marsh zone, but also be instrumental in changing the species composition of the marsh zone".

Water level increases were expected to occur in Kiskittogisu Lake also, resulting in shoreline flooding, particularly in the low-lying southern portion of the lake. Because Kiskitto Lake was to be regulated independently of Lake Winnipeg, its levels were not expected to fluctuate in the same manner as the other lakes. However, a rise in the lake's mean level, combined with reduced water fluctuations, were still expected to cause a substantial reduction of marsh areas.

Webb (1973) noted that fall staging use of Playgreen Lake by scaup and other ducks was moderately high in 1972. Shorelines along the south basin of Playgreen Lake were rated by the Canada Land Inventory as being important areas for migrating waterfowl (Hutchison and Adams 1970). Waterfowl were expected to continue to use the lake as a staging area in fall, but at reduced levels due to Lake Winnipeg regulation. Water level increases were expected to remove some waterfowl nesting habitat.

Moderate numbers of migrating tundra swans, scaup, mallards and Canada geese used Kiskittogisu Lake in fall, 1972. Hutchison and Adams (1970) identified a number of areas that served as important migration stops. Some birds were expected to be displaced as a result of habitat changes caused by modified flow regimes. Numerous nesting islands could also be inundated. Excavation of 8-Mile Channel and associated deposition of dredge spoils was expected to adversely affect the habitat of nesting waterfowl and sandhill cranes and lead to a reduction in use by those species. The channel was expected to carry additional sediment loads to Kiskittogisu Lake and increase its turbidity, which in turn could adversely affect the growth of submergent vegetation and limit the availability of food for staging waterfowl.

Hutchison and Adams (1970) considered parts of the lake to be important for staging waterfowl. The Study Board considered that regulation of Kiskitto Lake levels independent of West Channel flows would result in less frequent flooding of nesting islands in the lake and reduce the number of waterfowl nests flooded in the spring. However, if water levels were maintained within too narrow a range, stagnation and loss of marsh vegetation could occur and the lake's value to waterfowl would be reduced.

4.1.2 Study Board Recommendations

The Study Board made a number of recommendations that concerned the entire Nelson River hydroelectric development scheme, and others that were specific to the Outlet Lakes area (LWCNRSB 1975c).

Recommendations of general application that are relevant to the present study as follows:

> 5. *That a mechanism be established to deal with social and related and economic issues including:*
>
>   (c) *monitoring and analysis of ongoing social and economic changes related to hydroelectric development and more generally northern development.*
>
> 10. *That appropriate government departments and agencies develop and implement a long-term coordinated ecological monitoring and research program to allow impact evaluation and to assist in the ongoing management of the affected area.*

One recommendation, specific to the Outlet Lakes area, applies directly to the present study:

29. *That resource management and resource agencies investigate further the operational aspects of the Kiskitto Lake regulation works to determine optimum water levels for fish, wildlife and recreational resources.*

4.2 Canadian Wildlife Service Studies

4.2.1 Kiskittogisu River

The impact of construction of 8-Mile Channel on migratory bird use of Kiskittogisu River and adjacent habitat was assessed by Rakowski (1975).

Kiskittogisu River, including its lagoon system at the downstream end, was considered to provide good quality rearing habitat for ducks and geese and was regarded as important for staging and migrating waterfowl. This habitat was expected to be severely affected by sedimentation as a result of deposition of dredge spoils during construction of the 8-Mile Channel. Rakowski predicted that all available high-quality waterfowl habitat in the vicinity of Kiskittogisu River would be lost through flooding once the channel connected Playgreen and Kiskittogisu lakes.

4.2.2 Kiskitto Lake

In his report to the Study Board, Webb (1973) recommended that optimal water levels for Kiskitto Lake be determined to lessen the impact of Lake Winnipeg regulation on wildlife. Subsequently, Canadian Wildlife Service initiated a study to evaluate the current and potential value of Kiskitto Lake to waterfowl, and to recommend development and management practices that would benefit waterfowl.

Rakowski (1976) concluded that the lake was probably more important for waterfowl staging than production, and recommended that lake levels be maintained between 212.60 and 213.36 m a.s.l. The water level should be lowered in the spring and maintained at approximately 212.60 m a.s.l. until August to (a) create additional shoreline habitat for nesting waterfowl; (b) prevent flooding of waterfowl nests; and (c) maintain habitat in optimum condition. High water levels could be maintained every third or fourth year in order to control undesirable plant succession.

4.3 Kiskitto Lake Regulation Committee Study

The Kiskitto Lake Regulation Committee, comprised of representatives of the Manitoba government and Manitoba Hydro, was formed in August 1975 in response to Study Board Recommendation 29 (see subsection 4.1.2). The terms of reference of the Committee required it to "develop operating rules for the Kiskitto Lake control structure that will optimize the value of the resources associated with the lake, recognizing environmental and social considerations in addition to the economic value of the resources" (Kiskitto Lake Regulation Committee 1977).

The Committee recommended that the lake be regulated between 212.75 and 213.36 m a.s.l., and that a minimum inflow of 2.83 m^3/sec from the Nelson River West Channel be maintained throughout the year to prevent eutrophication. To improve flushing of the lake, inflow should be increased to full culvert capacity when water is being spilled at the Jenpeg Generating Station. However, lake levels should not be permitted to exceed the 213.36m upper elevation to prevent flooding of muskeg.

5. METHODS

5.1 Impact Assessment and Impact Management Approach

The flowchart in Figure 2 illustrates the approach used in determining how the waterfowl-related interests and opportunities of Norway House Band Indians in the fall have been affected by the LWR project and, consequently, what compensation is required by the **NFA**. References are made in the flowchart to the pertinent recommendations of the Study Board and provisions of the **NFA** which are addressed by the process. Reference to "adverse" effects or impacts in this report reflects terminology of the **NFA** and does not represent any preconclusion of this study.

The assessment process begins with identifying the effects of LWR on waterfowl use of the Outlet Lakes area in fall (biophysical impacts). The direct, indirect, or cumulative consequences of these biophysical impacts are expected to affect the economic, cultural, and social life of Norway House Band people.

5.1.1 Biophysical Impacts

Determining how the LWR project has affected waterfowl use of the Outlet Lakes area partially fulfills Recommendation 10 of the Study Board and **NFA** sub-article 17.5 (See Appendix A).

Recommendation 10 recognizes that ongoing management of the Outlet Lakes area requires that information first be gathered on how the area has been impacted by development. The present study focuses on how use of the Outlet Lakes by fall-staging waterfowl has been affected by the Lake Winnipeg regulation.

NFA sub-article 17.5 (of Article 17 - Environmental Impact Policy) requires that monitoring of effects of the project be carried out so as to "provide such information as may be necessary to give effect" to the **NFA** (see Appendix A). Giving effect to the **NFA** means ensuring that all members of the five **NFA**

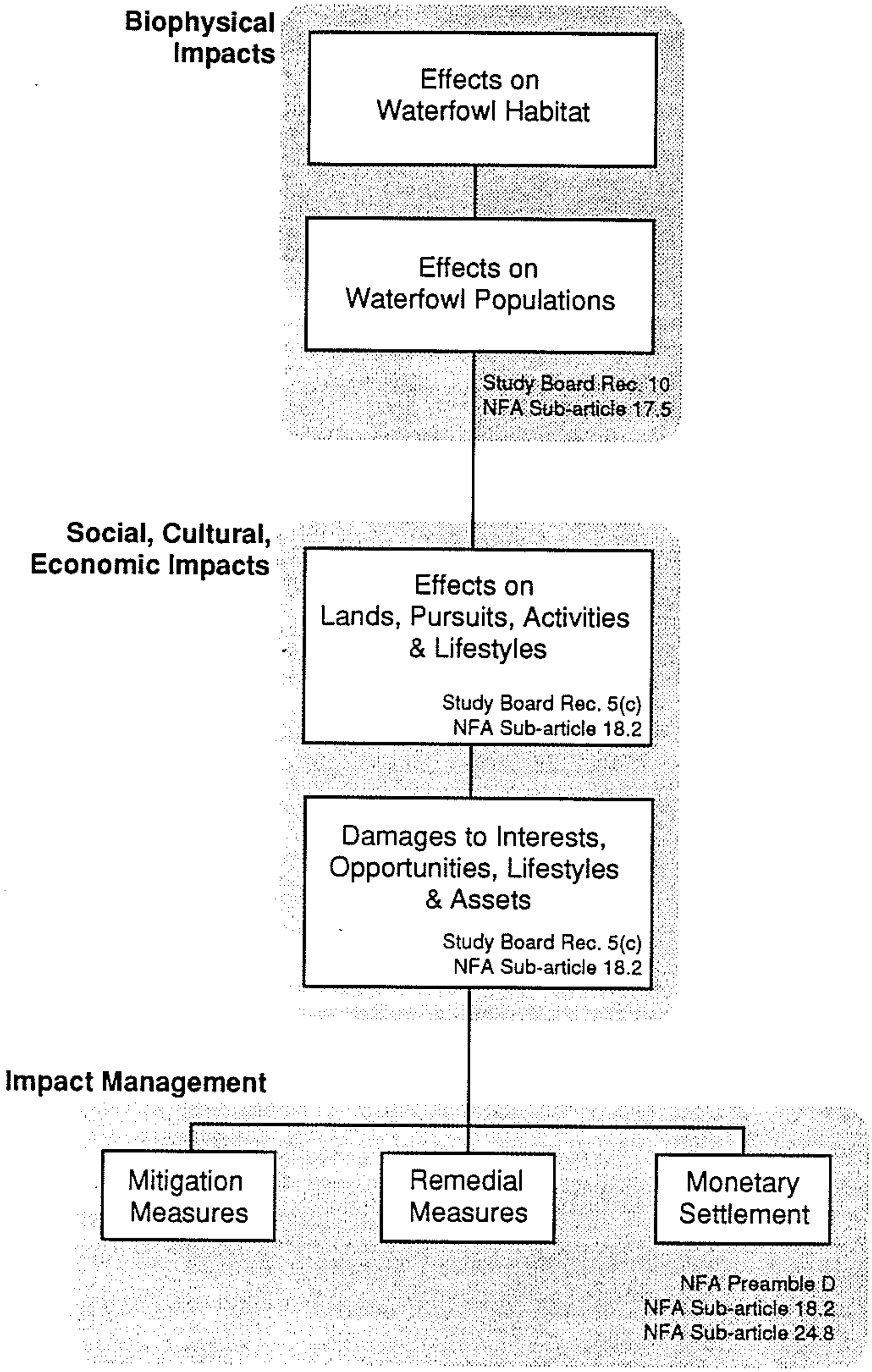

Figure 2. The impact assessment and impact management approach used in the study.

who have been or may be adversely affected by the hydroelectric development, are dealt with or compensated "fairly and equitably" (Preamble D to the **NFA**).

Collection of data that document the effects of LWR on waterfowl use of the Outlet Lakes constitutes compliance with sub-article 17.5 with respect to the waterfowl-related impacts that the Norway House claims to have experience. In addition, documentation of effects on waterfowl in the Outlet Lakes area represents a response, in part, to Part III of Claim 18, which embodies the intent and meaning of Recommendation 10 and **NFA** sub-article 17.5. Claim 18 is described in Appendix A.

5.1.2 Socio-Cultural-Economic Impacts

Determining how LWR has affected the waterfowl-related interests of Norway House Band people partially fulfills Study Board Recommendation 5(c) and **NFA** sub-article 18.2.

Recommendation 5(c) called for "monitoring and analysis of ongoing social and economic changes related to hydroelectric development" (subsection 4.1.2 of this report). The present study addresses those changes experienced by Norway House Band people that are associated with effects on waterfowl use of the Outlet Lakes area, i.e., changes in waterfowl hunting opportunities.

Collection of information that documents the effects of LWR on the waterfowl hunting opportunities of Norway House Band people constitutes compliance with **NFA** sub-article 18.2 with respect to the waterfowl-related impacts experienced by the Norway House Band. In addition, this documentation represents a response, in part, to Part II of Claim 18, which embodies the intent and meaning of Recommendation 5(c) and **NFA** sub-article 18.2. Finally, information dealing with adverse effects on waterfowl hunting opportunities allows damages to the waterfowl-related interests, opportunities, lifestyles and assets of Norway House people to be defined.

5.1.3 Impact Management

Impact Management refers to a mechanism, or set of mechanisms, by which adverse impacts of a project may be managed, whether they be biophysical (e.g., directly affecting waterfowl) or socio-cultural-economic impacts (e.g., affecting waterfowl hunting opportunities). Suitable and adequate compensation for adverse effects on **NFA** Bands would constitute redress for damages suffered by **NFA** Bands. **NFA** sub-article 24.8 refers to three forms of compensation or redress for damages suffered: mitigating measures, remedial measures and monetary settlement.

5.2 Data Collection

5.2.1 Effects on Waterfowl and Habitat

On June 20, 1986, an aerial reconnaissance of the 8-Mile Channel and Kiskittogisu River area was carried out using a Bell 206B helicopter to determine general habitat conditions following construction of 8-Mile Channel. Additional observations on shoreline habitat were made during the aerial surveys of waterfowl populations conducted in fall, 1986 and 1987.

Fall staging of waterfowl in the Outlet Lakes area was determined through a series of low-altitude aerial surveys conducted in 1986 and 1987. The 1986 surveys were flown on September 5 and 12 in a Cessna 185 with two observers; the October 5, 16 and 26, 1987 surveys were flown in a Bell 206B helicopter with one observer. All Surveys were flown at altitudes of between 30 and 40 m above ground level. Survey routes followed shorelines of Playgreen, Kiskitto and Kiskittogisu lakes and 8-Mile Channel (Figure 3). Table 1 gives the length of each survey in each lake.

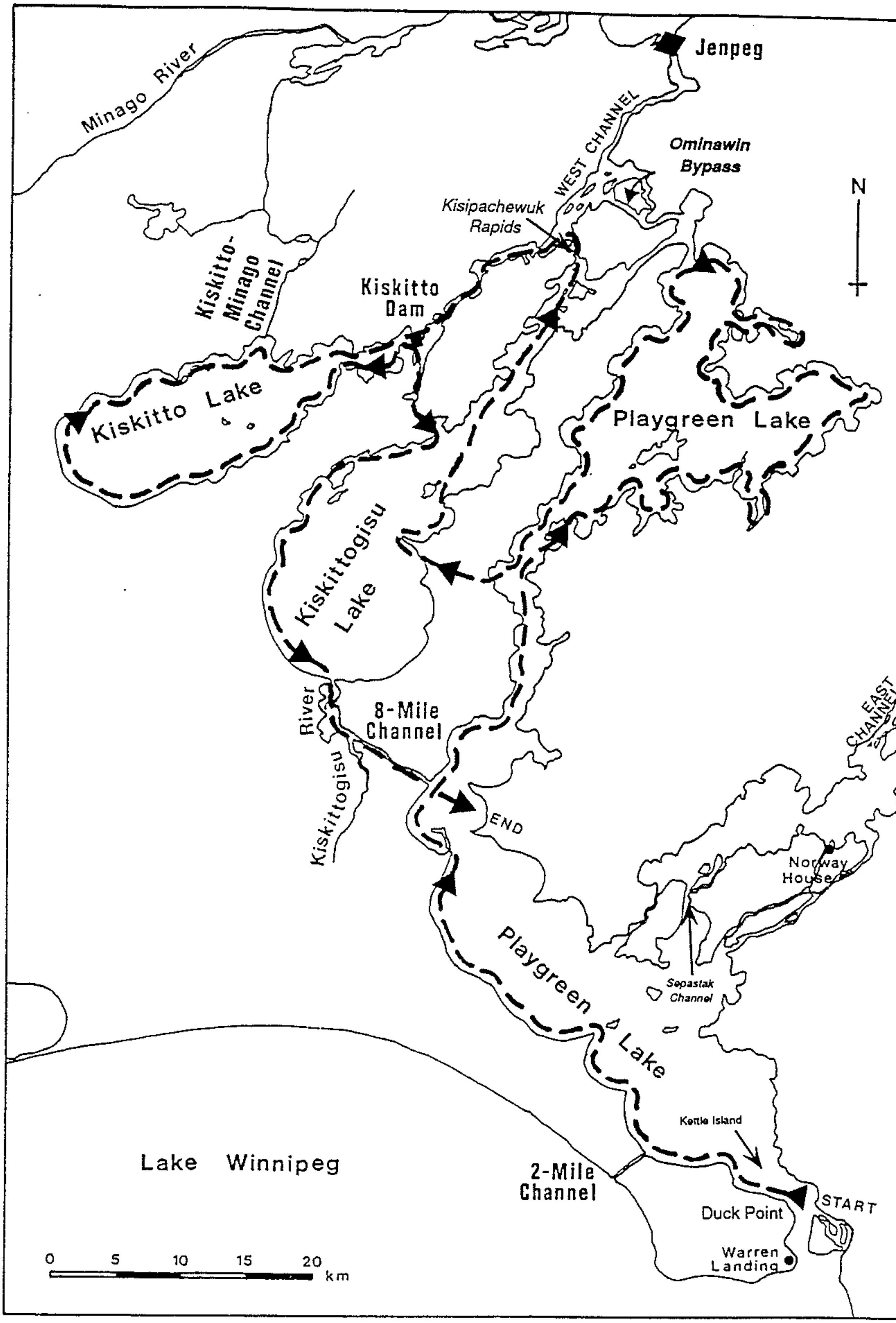

Figure 3. Waterfowl survey route, 1986/87.

Table 1. Lengths (km) of post-regulation surveys of the Outlet Lakes area.

Area	Sept. 5, 1986	Sept. 12, 1986	Oct. 5, 1987	Oct. 16, 1987	Oct. 26, 1987
Playgreen-South Basin	59.0	59.0	137.0	145.0	133.0
Playgreen-North Basin	151.0	151.0	151.0	145.5	145.5
Kiskitto Lake	85.0	85.0	85.0	30.8	n.s.
Kiskittogisu Lake	101.0	101.0	101.0	12.0	13.5
TOTAL	**396.0**	**396.0**	**474.0**	**333.3**	**292.0**

n.s. = not surveyed

5.2.2 Effects on Norway House Band

On February 13, 1987, one-on-one interviews were held with seven waterfowl hunters in Norway House. Hunters were asked to describe: their waterfowl hunting experiences before and after Lake Winnipeg regulation; traditional hunting locations used prior to regulation; hunting success at traditional hunting locations before and after regulation; and species of waterfowl hunted. They were also asked to share their observations on changes in numbers and distribution of waterfowl and their perceptions on the causes of these changes and the influence of Lake Winnipeg regulation. Because the purpose of the study was to investigate the effects of the diversion on fall waterfowl use and fall hunting, discussions focussed on the fall hunting period.

6. RESULTS

6.1 General Habitat Conditions

Habitat conditions along Kiskittogisu River and in the vicinity of 8-Mile Channel were examined by helicopter on June 20, 1986. Figure 4 shows how 8-Mile Channel intercepts the former natural outlet of Kiskittogisu River before entering Kiskittogisu Lake.

8-Mile Channel has caused the level of Kiskittogisu River to exceed its natural capacity and flood vegetation that previously grew along its banks. Remains of dead white birch (*Betula papyrifera*), black spruce (*Picea mariana*) and other trees occur along the river course.

Although no ground inspection was carried out, the islands created by depositing dredge spoils along 8-Mile Channel have become well-vegetated with willow (*Salix*) and various aquatic plants. Where Kiskittogisu River enters 8-Mile Channel appears to have become stabilized with aquatic vegetation.

6.2 Waterfowl Use

Data on waterfowl use have been summarized for the following survey areas: Playgreen Lake north basin, Playgreen Lake south basin, Kiskitto Lake and Kiskittogisu Lake (Figure 1).

The survey results presented here focus primarily on numbers of birds observed per kilometre surveyed. Because survey dates are widely separated both within and between years, and represent different observation "platforms" (fixed-wing and helicopter aircraft), no attempt has been to calculate average densities. Instead, discussion focuses on the highest densities recorded because they are more likely to represent the potential for waterfowl use. Highest densities are also used in the comparison of pre- and post-regulation use of the Outlet Lakes by waterfowl (see Section 7).

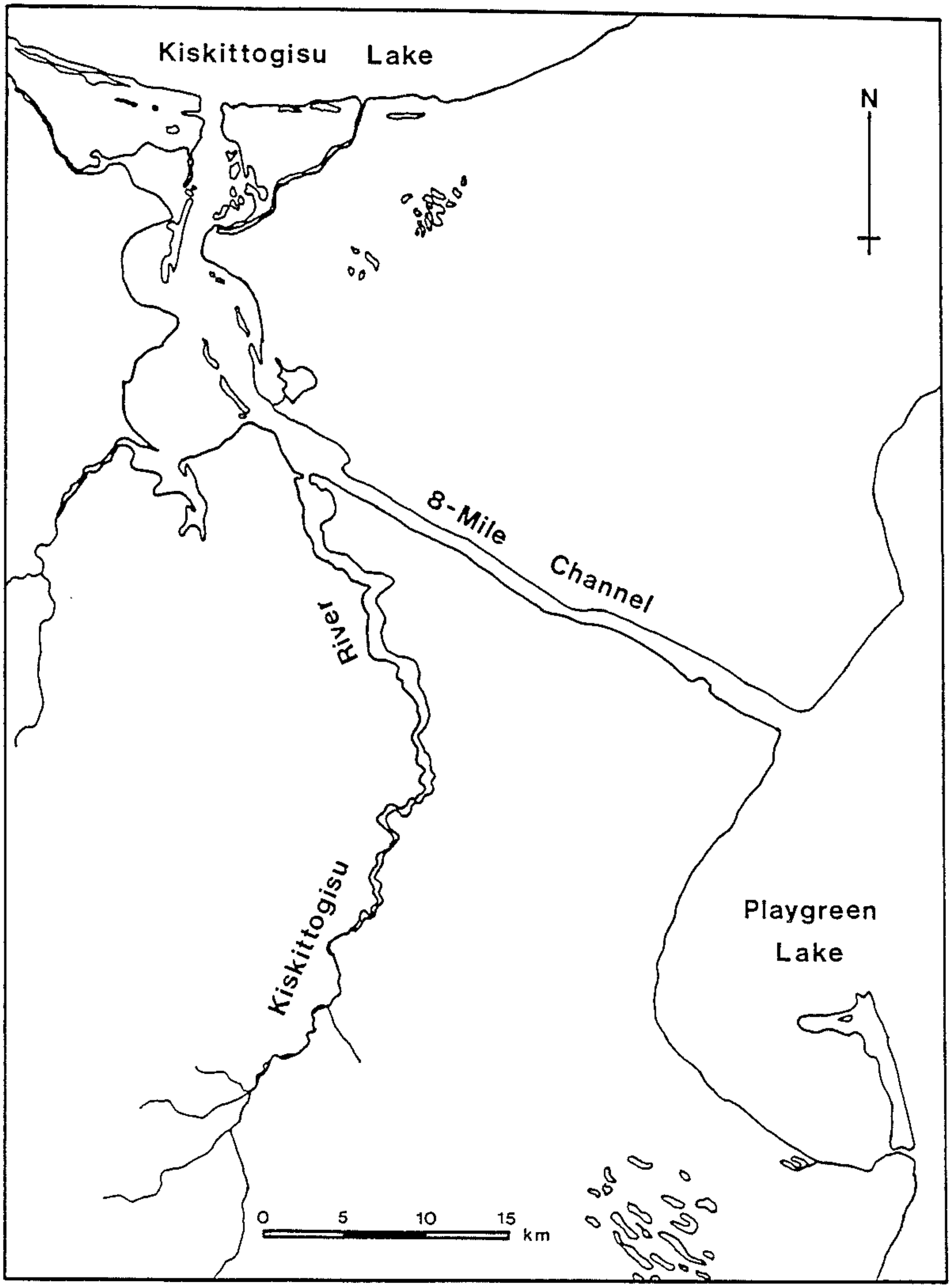

Figure 4. Location of 8-mile channel in relation to Kiskittogisu River, Kiskittogisu Lake, and Playgreen Lake (south basin).

The September and October survey dates were chosen to: (a) correspond with pre-regulation surveys carried out in 1972 and 1973, and (b) coincide with major waterfowl movements based on long-term experience of Norway House residents.

Diving Ducks

Playgreen Lake received the most use by diving ducks in 1986/87 with the north basin attracting more birds per kilometre of shoreline than the south basin (Figure 5). Lesser scaup (*Aythya affinis*) was the most abundant diver species while common goldeneye (*Bucephala clangula*), bufflehead (*Bucephala albeola*) and scoters (*Melanitta* spp.) were observed in lower numbers.

Highest densities of diving ducks recorded during any survey segment of the Outlet Lakes ranged from .13 birds/km at Kiskitto Lake to 2.69 birds/km at Playgreen Lake, North Basin; densities at Playgreen Lake, South basin were almost as high -- 2.37 birds/km (Table 2, Figure 5). Densities were consistently higher in 1987 than in 1986, which probably reflects the difference in survey techniques for the two years (fixed-wing aircraft in 1986; helicopter in 1987). Two of the Kiskitto Lake surveys and two of the Kiskittogisu Lake surveys yielded no diving ducks.

Dabbling Ducks

Dabbling duck use of Playgreen Lake was low in 1986 (Table 3, Figure 5). Mallards (*Anas platyrhynchos*) were the most common frequently observed species, with gadwall (*Anas strepera*) and American wigeon (*Anas americana*) present in lower numbers.

Highest densities of dabbling ducks during individual survey segments varied from 0.30 birds/km (Kiskitto Lake) to 4.58 birds/km (Kiskittogisu Lake). The highest density recorded at Playgreen Lake, North Basin was 3.06 birds/km, versus 2.50 recorded in the South Basin. Ducks were observed on all but one survey (September 12, 1986) of the two Playgreen Lake segments; however, at

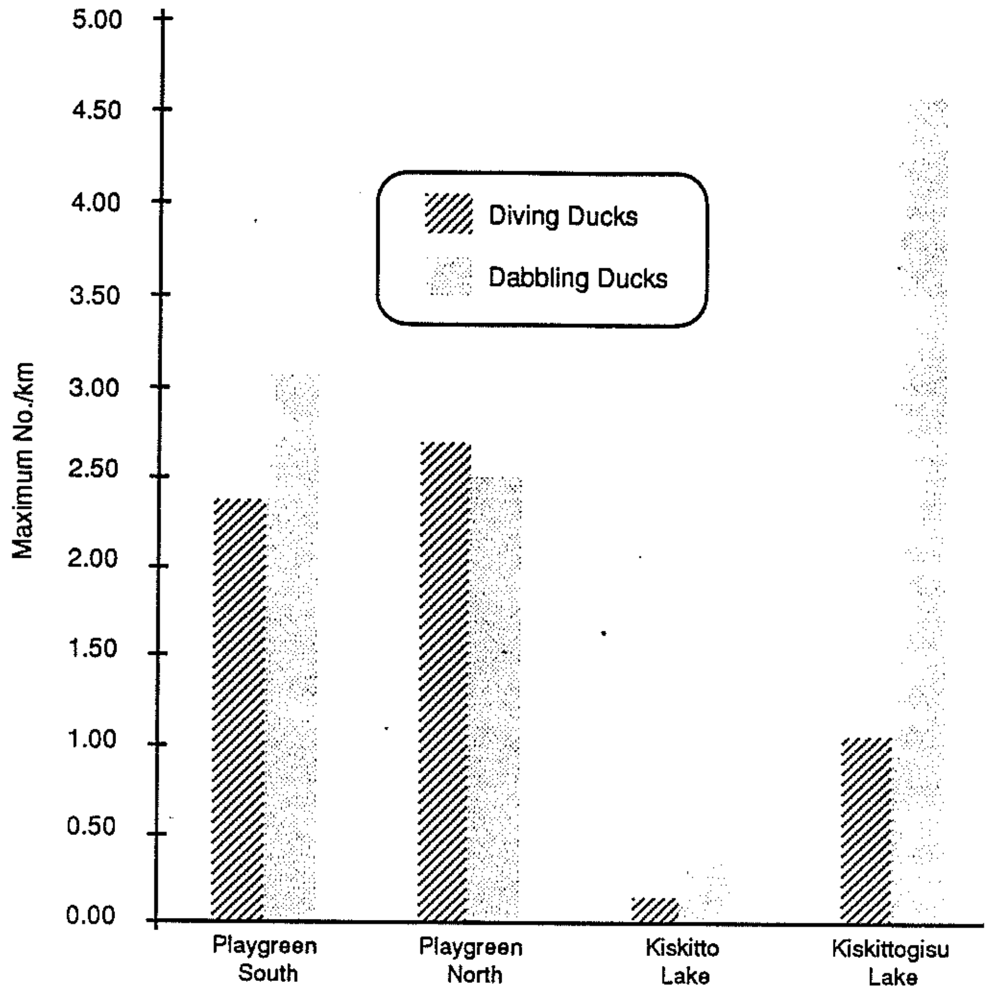

Figure 5. Diving and dabbling duck use of the Outlet Lakes area in Fall, 1986/87.

Table 2. Numbers and densities* of diving ducks observed during post-regulation surveys of the Outlet Lakes area.

Area	5 Sept. 1986 no.	5 Sept. 1986 density	12 Sept. 1986 no.	12 Sept. 1986 density	5 Oct. 1987 no.	5 Oct. 1987 density	16 Oct. 1987 no.	16 Oct. 1987 density	26 Oct. 1987 no.	26 Oct. 1987 density	Highest Density
Playgreen–South Basin	0	0.00	4	0.07	92	0.67	212	1.46	315	2.37	2.37
Playgreen–North Basin	309	2.05	99	0.66	406	2.69	308	2.12	287	1.97	2.69
Kiskitto Lake	10	0.12	0	0.00	0	0.00	4	0.13	n.s.	–	0.13
Kiskittogisu Lake	25	0.25	33	0.33	105	1.04	0	0.00	0	0.00	1.04
TOTAL & AVG. DENSITY	**344**	**0.87****	**136**	**0.34****	**603**	**1.27****	**524**	**1.57****	**602**	**2.06****	

* birds/km
**Based on total survey length: 396 km (5 & 12 Sept.); 474 km (5 Oct.); 333.3 km (16 Oct.); 292 km (26 Oct.)
n.s. = not surveyed

Table 3. Numbers and densities* of dabbling ducks observed during post-regulation surveys of the Outlet Lakes area.

Area	5 Sept. 1986 no.	5 Sept. 1986 density	12 Sept. 1986 no.	12 Sept. 1986 density	5 Oct. 1987 no.	5 Oct. 1987 density	16 Oct. 1987 no.	16 Oct. 1987 density	26 Oct. 1987 no.	26 Oct. 1987 density	Highest Density
Playgreen–South Basin	2	0.03	0	0.00	163	1.19	444	3.06	32	0.24	3.06
Playgreen–North Basin	91	0.60	16	0.11	103	0.68	364	2.50	14	0.10	2.50
Kiskitto Lake	0	0.00	0	0.00	25	0.30	0	0.00	n.s.	–	0.30
Kiskittogisu Lake	30	0.30	0	0.00	137	1.36	55	4.58	0	0.00	4.58
TOTAL & AVG. DENSITY	**123**	**0.31****	**16**	**0.04****	**428**	**0.9****	**863**	**2.59**	**46**	**0.16****	

* birds/km
**based on total survey length: 396 km (5 & 12 Sept.); 474 km (5 Oct.); 333.3 km (16 Oct.); 292 km (26 Oct.)
n.s. = not surveyed

Kiskitto Lake birds were observed on only one of the surveys (October 5, 1987). The high density recorded at Kiskittogisu Lake (October 16, 1987) is based on a short survey segment (12 km) and may represent a biased estimate of waterfowl use for the entire lake basin.

Ducks (Dabbling, Diving, and Unidentified species)

Highest densities for all ducks (Table 4, Figure 6) were about the same for Playgreen Lake North (4.86 birds/km) and South (4.62 birds/km) Basins and for Kiskittogisu Lake (4.58 birds/km). Given the range of time covered by the surveys -- early September to late October over two seasons -- the densities for Playgreen North and, to a lesser extent, Playgreen South are surprisingly consistent. Results also suggest that the October 16 survey best represented ducks present in the Outlet Lakes region in Fall, 1987. The generally low densities recorded in 1986 may indicate surveys that year occurred before large numbers of migrants had reached the study area.

Canada Geese

Numbers and densities of Canada geese were low through all survey dates and segments (Table 5, Figure 6), with highest densities ranging from 0.19 birds/km (Playgreen Lake North Basin) to 1.46 birds/km (Playgreen Lake South Basin). The largest number of geese recorded on any survey segment/date was 54.

6.3 Hunter Interviews

Traditionally, Norway House Band members have hunted waterfowl during spring and fall migrations. Canada geese are hunted almost exclusively in spring, whereas ducks are hunted during both seasons.

Waterfowl hunters expressed their concerns about the effects of diversion during interviews conducted as part of this study (Figure 7). Hunters felt that the availability of ducks and geese at traditional hunting sites is considerably lower than it was before LWR. As a consequence, hunters were returning home with

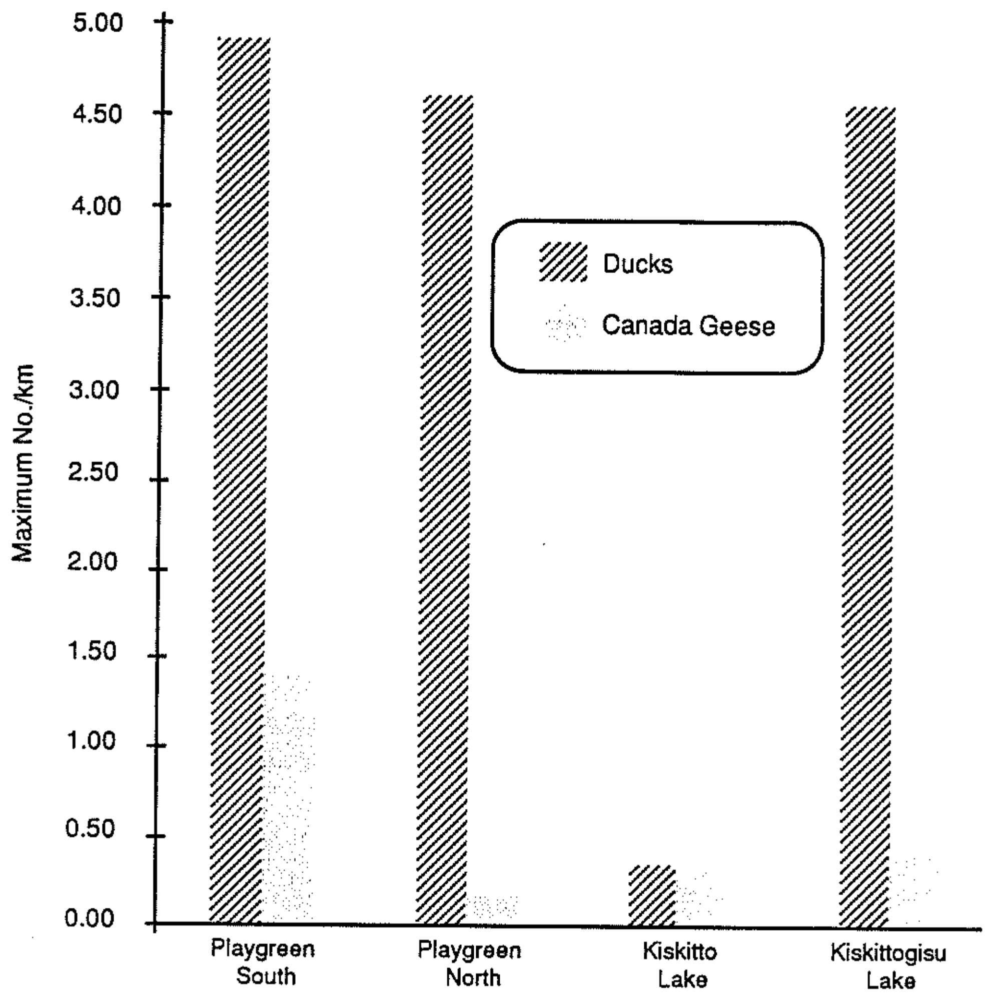

Figure 6. Use of the Outlet Lakes area by ducks (divers, dabblers, unidentified) and Canada geese in Fall, 1986/87.

WATERFOWL-RELATED CONCERNS AND ADVERSE EFFECTS

Observations of Norway House Band Hunters*

Habitat

- pondweeds are not as numerous as before
- pondweeds seem to be drowned out
- Playgreen Lake has become murky
- there are concerns about mercury levels and organic material in the lake

Waterfowl

- ducks generally are not as numerous as before regulation
- dùcks are not as numerous in the Warren Landing narrows
- there has been a noticeable drop in duck numbers in the Sepastak Channel area
- ducks tend to raft in Playgreen Lake away from shore; they used to be distributed in shallow bays close to shore
- there has been no noticeable change in numbers of ducks and Canada geese in spring
- reductions in duck numbers have occurred since 2-mile Channel and the transmission line were installed
- most ducks now go as far as Kettle Island or Duck Point then rise to a higher altitude; they used carry on in low flights to Warren Landing Narrows

Hunting

- there has been little success in hunting at traditional sites; returns have been poor for the effort spent
- it used to be easy to decoy ducks to Kettle Island before regulation
- there is concern over the size of the hunter kill in the United States
- hunters used to be able to rely on a succesful hunt near Warren Landing and along the east shore of Playgreen Lake
- Kiskittogisu River used to be good for geese in spring and fall
- hunters used to be able to bag about 10-15 ducks an hour; four years ago, they bagged only 3 ducks over three evenings

*Comments listed here represent the opinions of individual hunters

Figure 7. Waterfowl-related concerns and adverse effects identified during hunter interviews at Norway House.

Table 4. Numbers and densities* of ducks^ observed during post-regulation surveys of the Outlet Lakes area.

Area	5 Sept. 1986		12 Sept. 1986		5 Oct. 1987		16 Oct. 1987		26 Oct. 1987		Highest Density
	no.	density	no.	density	no.	density	no.	density	no.	density	
Playgreen–South Basin	2	0.03	4	0.07	310	2.26	705	4.86	382	2.37	4.86
Playgreen–North Basin	400	2.65	107	0.71	559	3.70	672	4.62	301	2.07	4.62
Kiskitto Lake	10	0.12	0	0.00	25	0.30	4	0.13	n.s.	–	0.30
Kiskittogisu Lake	55	0.55	33	0.33	242	2.40	55	4.58	0	0.00	4.58
TOTAL & AVG. DENSITY	**467**	**1.18****	**144**	**0.36****	**1136**	**2.4****	**1436**	**4.31****	**683**	**2.34****	

* birds/km
^includes dabblers, divers, and unidentified ducks
**based on total survey length: 396 km (5 & 12 Sept.); 474 km (5 Oct.); 333.3 km (16 Oct.); 292 km (26 Oct.)
n.s. = not surveyed

Table 5. Numbers and densities* of Canada geese observed during post-regulation surveys of the Outlet Lakes area.

Area	5 Sept. 1986		12 Sept. 1986		5 Oct. 1987		16 Oct. 1987		26 Oct. 1987		Highest Density
	no.	density	no.	density	no.	density	no.	density	no.	density	
Playgreen–South Basin	0	0.00	0	0.00	28	0.67	17	1.46	0	0.00	1.46
Playgreen–North Basin	15	0.10	29	0.19	17	0.11	23	0.16	0	0.00	0.19
Kiskitto Lake	8	0.10	54	0.64	0	0.00	0	0.00	n.s.	–	0.64
Kiskittogisu Lake	38	0.38	15	0.15	42	0.42	0	0.00	0	0.00	0.42
TOTAL & AVG. DENSITY	61	**0.15****	98	**0.25****	87	**0.18****	40	**0.12****	0	**0.00**	

* birds/km
**based on total survey length: 396 km (5 & 12 Sept.); 474 km (5 Oct.); 333.3 km (16 Oct.); 292 km (26 Oct.)
n.s. = not surveyed

substantially fewer ducks per unit effort than they harvested before regulation. In contrast, spring waterfowl hunting opportunities appeared to remain largely unchanged.

Figure 8 shows the areas frequently used for hunting ducks in the fall prior to LWR. Of these, the group of islands in the vicinity of Warren Landing received the most use. Hunters described how, in the past, they observed flocks of ducks flying southeast in the fall across Playgreen Lake to the Warren Landing area en route to Lake Winnipeg and points south. Hunters would set up blinds on the shores of the islands east of Warren Landing and decoy ducks into shallow bays where they could be shot and retrieved relatively easily.

Hunters complained that since 2-Mile Channel was constructed and Lake Winnipeg regulation began, ducks no longer visit the Warren Landing area in large numbers. Consequently, hunters get very few geese for their hunting effort. Some hunters observed ducks deviating from their traditional flight path towards Warren Landing and using 2-Mile Channel to gain access to Lake Winnipeg. They indicated that the deep and fast-flowing waters of 2-Mile Channel are not suitable for hunting with decoys. Other hunters had seen flocks travel as far as Kettle Island or Duck Point (Figure 8) and raft offshore in waters too deep to permit recovery of birds. Similarly, hunters reported noticeable reductions in the number of ducks in the "Sepastak" area, and in the bays along the eastern shoreline of Playgreen Lake.

Some hunters noted physical changes in Playgreen Lake and claimed the lake was now more turbid.

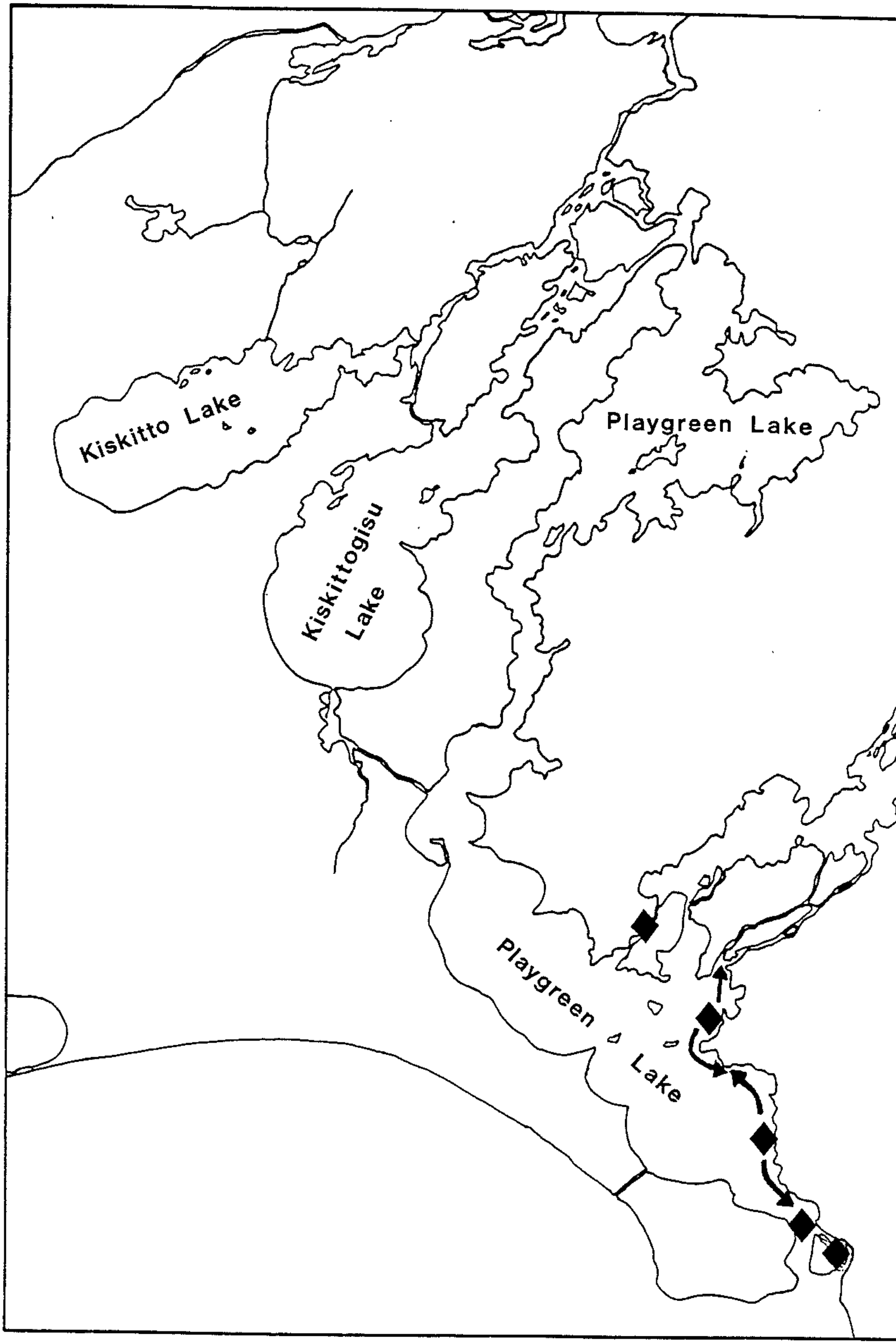

Figure 8. Traditional fall waterfowl hunting areas of the Norway House Band in the Outlet Lakes area.

7. DISCUSSION

7.1 Pre- and Post-Regulation Waterfowl Use

Aerial surveys of shorelines in the Outlet Lakes area were carried out before Lake Winnipeg regulation by Webb (1973) on September 24 and 25, 1972 and by Manitoba government biologists on August 30, September 7, September 14 and October 6, 1973 (Rakowski 1976). Table 6 gives the length of each survey in each lake.

Diving Ducks

Comparison of 1972/73 and 1986/87 survey results shows densities of diving ducks in the two Playgreen Lake basins to be substantially higher before LWR (Table 7) than after (Table 2). Highest densities of 72.07 and 102.01 birds/km recorded in Playgreen Lake north and south basins before regulation were 30 to 40 times greater than highest densities (2.37 and 2.69 birds/km) reported after regulation.

Densities of diving ducks at Kiskitto and Kiskittogisu lakes were also greater before LWR, although the data for Kiskitto Lake are too fragmentary for firm conclusions to be drawn. The highest density at Kiskittogisu Lake in 1972/73 (4.83 birds/km) was about 5 times greater than that recorded in 1986/87 (1.04 birds/km).

Dabbling Ducks

Comparison of pre-and post-regulation survey results (Tables 8 and 3) for dabblers is inconclusive. In 1972/73, dabbling ducks were observed on Playgreen Lake South Basin on three of five surveys, and on the North Basin on only two of five surveys. In 1986/87, birds were consistently observed on Playgreen Lake through the entire period surveyed. The highest density recorded for South Basin were about 2 times greater in 1986/87 than in 1972/73 (3.06 versus 1.49 birds/km). In North Basin, the highest pre-regulation density was 3 time greater than the highest post-regulation density (7.87 versus 2.5 birds/km). Overall, there

Table 6. Lengths (km) of pre-regulation surveys of the Outlet Lakes area.

Area	Sept. 24/25 1972	Aug. 30 1973	Sept. 7 1973	Sept. 14 1973	Oct. 6 1973
Playgreen-South Basin	113.5	88.8	88.8	19.5	59.0
Playgreen-North Basin	71.8	49.8	n.s.	94.0	n.s.
Kiskitto Lake	85.8	36.5	57.0	27.4	84.0
Kiskittogisu Lake	154.8	45.1	54.6	45.0	92.5
TOTAL	425.9	220.2	200.4	185.9	235.5

n.s. = not surveyed

Table 7. **Numbers and densities* of diving ducks observed during pre-regulation surveys of the Outlet Lakes area.**

Area	24/25 Sept. 1972		30 Aug. 1973		7 Sept. 1973		14 Sept. 1973		6 Oct. 1973		Highest Density
	no.	density	no.	density	no.	density	no.	density	no.	density	
Playgreen–South Basin	2930	25.81	4135	46.57	6400	72.07	277	14.21	1505	25.51	72.07
Playgreen–North Basin	4534	63.15	5080	102.01	n.s.	n.s	770	8.19	n.s.	n.s.	102.01
Kiskitto Lake	174	2.03	0	0.00	0	0.00	0	0.00	0	0.00	2.03
Kiskittogisu Lake	747	4.83	85	1.88	0	0.00	0	0.00	300	3.24	4.83
TOTAL & AVG. DENSITY	8385	19.69**	9300	41.85**	6400	31.94**	1047	5.63**	1805	7.66**	

* birds/km
**Based on total survey length: 425.9 km (24/25 Sept.); 220.2 km (30 Aug.); 200.4 km (7 Sept.); 185.9 km (14 Sept.); 235.5 (6 Oct).
n.s. = not surveyed

Table 8. **Numbers and densities* of dabbling ducks observed during pre-regulation surveys of the Outlet Lakes area.**

Area	24/25 Sept. 1972		30 Aug. 1973		7 Sept. 1973		14 Sept. 1973		6 Oct. 1973		Highest Density
	no.	density	no.	density	no.	density	no.	density	no.	density	
Playgreen–South Basin	169	1.49	0	0.00	10	0.11	0	0.00	50	0.85	1.49
Playgreen–North Basin	565	7.87	10	0.20	0	0.00	0	0.00	n.s.	n.s.	7.87
Kiskitto Lake	77	0.90	0	0.00	0	0.00	21	0.77	0	0.00	0.90
Kiskittogisu Lake	88	0.57	85	1.88	0	0.00	21	0.47	0	0.00	1.88
TOTAL & AVG. DENSITY	899	2.11**	95	0.43**	10	0.05**	42	0.23**	50	0.21**	

* birds/km
**Based on total survey length: 425.9 km (24/25 Sept.); 220.2 km (30 Aug.); 200.4 km (7 Sept.); 185.9 km (14 Sept.); 235.5 (6 Oct).
n.s. = not surveyed

is no indication that use by dabbling ducks has changed since LWR. The survey results do suggest that the Outlet Lakes did not attract large numbers of dabblers, with or without regulation.

As noted for diving ducks, not enough dabbling ducks were observed on Kiskitto Lake surveys to determine the effects of LWR on duck use or populations trends.

Surveys at Kiskittogisu Lake suggest that dabbling duck densities may have increased slightly since regulation; however, the counts are too small and inconsistent both years to draw reliable conclusions. Furthermore, the 1986/87 highest density of 4.58 birds/km is suspect, given the short distance surveyed (12 km).

Ducks (Dabbling, Diving, Unidentified)

As one might expect, the diving duck counts dominate the total-duck picture, which suggests that duck use has declined significantly on Playgreen Lake since LWR (Tables 4 and 9). There is some indication that densities at Kiskittogisu Lake have also declined since LWR, based on a highest density of 8.39 birds/km in 1972/73 versus 4.58 birds/km in 1986/87.

Inclusion of unidentified ducks to the survey results shows total duck densities at Kiskitto Lake to be higher before LWR than after, although numbers again are too small and inconsistent to be conclusive.

Canada Geese

Densities of Canada geese on Playgreen Lake North Basin, Kiskitto Lake, and Kiskittogisu Lake in 1972/73 were 2.6 and 4.1 times greater than in 1986/87, based on highest densities recorded (Tables 10 and 5). The opposite was true for Playgreen Lake South Basin, where goose densities were 7.3 times higher in 1986/87. Given the relatively small numbers of birds observed on individual surveys (12 to 175 in 1972/73; 8 to 42 in 1986/87) and the fact that no geese were

Table 9. Numbers and densities* of ducks^ observed during pre-regulation surveys of the Outlet Lakes area.

Area	24/25 Sept. 1972		30 Aug. 1973		7 Sept. 1973		14 Sept. 1973		6 Oct. 1973		Highest
	no.	density	no.	density	no.	density	no.	density	no.	density	Density
Playgreen–South Basin	3315	29.21	4225	47.58	6410	72.18	277	14.21	1555	26.36	72.18
Playgreen–North Basin	5099	71.02	5190	104.22	n.s.	n.s	800	8.51	n.s.	n.s.	104.22
Kiskitto Lake	341	3.98	20	0.55	0	0.00	21	0.77	0	0.00	3.98
Kiskittogisu Lake	1299	8.39	145	3.22	0	0.00	21	0.47	300	3.24	8.39
TOTAL & AVG. DENSITY	**10054**	**23.61****	**9580**	**43.51****	**6410**	**31.99****	**1119**	**6.02****	**1855**	**7.66****	

* birds/km
^includes dabblers, divers, and unidentified ducks.
**Based on total survey length: 425.9 km (24/25 Sept.); 220.2 km (30 Aug.); 200.4 km (7 Sept.); 185.9 km (14 Sept.); 235.5 (6 Oct).
n.s. = not surveyed

Table 10. Numbers and densities* of Canada geese observed during pre-regulation surveys of the Outlet Lakes area.

Area	24/25 Sept. 1972		30 Aug. 1973		7 Sept. 1973		14 Sept. 1973		6 Oct. 1973		Highest
	no.	density	no.	density	no.	density	no.	density	no.	density	Density
Playgreen–South Basin	0	0.00	10	0.11	18	0.20	0	0.00	12	0.20	0.20
Playgreen–North Basin	56	0.78	22	0.44	n.s.	n.s	13	0.14	n.s.	n.s.	0.78
Kiskitto Lake	87	1.01	97	2.66	12	0.21	0	0.00	0	0.00	2.66
Kiskittogisu Lake	175	1.13	12	0.27	12	0.22	23	0.51	42	0.45	1.13
TOTAL & AVG. DENSITY	**318**	**0.71****	**141**	**0.52****	**42**	**0.21****	**36**	**0.18****	**54**	**0.23****	

* birds/km
**Based on total survey length: 425.9 km (24/25 Sept.); 220.2 km (30 Aug.); 200.4 km (7 Sept.); 185.9 km (14 Sept.); 235.5 (6 Oct).
n.s. = not surveyed

observed on several surveys both before and after regulation, it is impossible to say whether the data represent any real change in goose use since LWR.

Figures 9 to 12 and Table 11 show the percentage change in diving and total duck numbers that occurred from 1972/73 (pre-regulation) to 1986/87 (post-regulation) for each of the four survey areas. Data for dabbling ducks and Canada geese are not included in Table 11 summary because the survey results are insufficient to make meaningful comparisons between pre- and post-regulation use by those species.

Results indicate that diving ducks declined by about 97% in Playgreen Lake and 78.5% in Kiskittogisu Lake between pre-and post-regulation periods. While comparisons are included for Kiskitto Lake, they must be considered inconclusive, owing to the limited amount of survey data available.

Considering the Outlet Lakes as a whole (Table 12), one can conclude that use by diving ducks has declined substantially (-96.2%) since LWR. The large decline (-90.1%) indicated for all ducks largely reflects the large diving duck component.

In general, it is risky to draw broad conclusions about changes in animal numbers from two periods of survey data represented by a limited number of "point-in-time" counts. Nevertheless, in the case of diving ducks, the between-year differences large enough to suggest use of the Playgreen Lake segment of the Outlet Lake system by divers has declined following regulation.

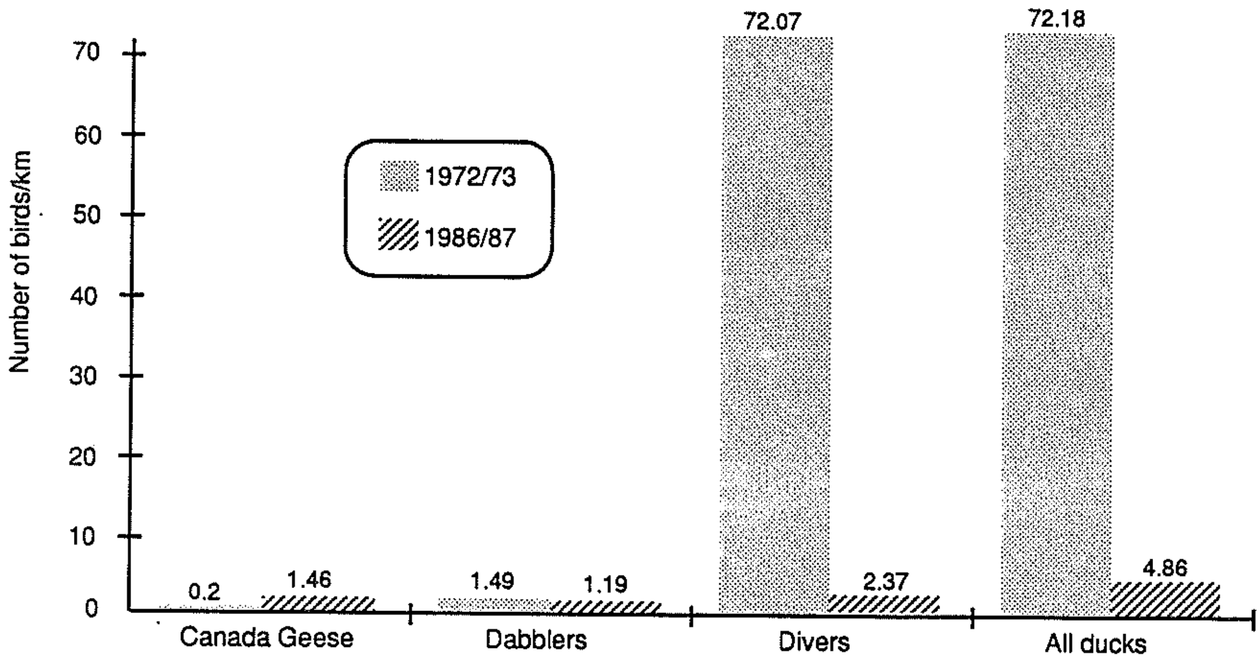

Figure 9. Pre- and post-regulation use of Playgreen–South Basin area by waterfowl based on highest survey densities.

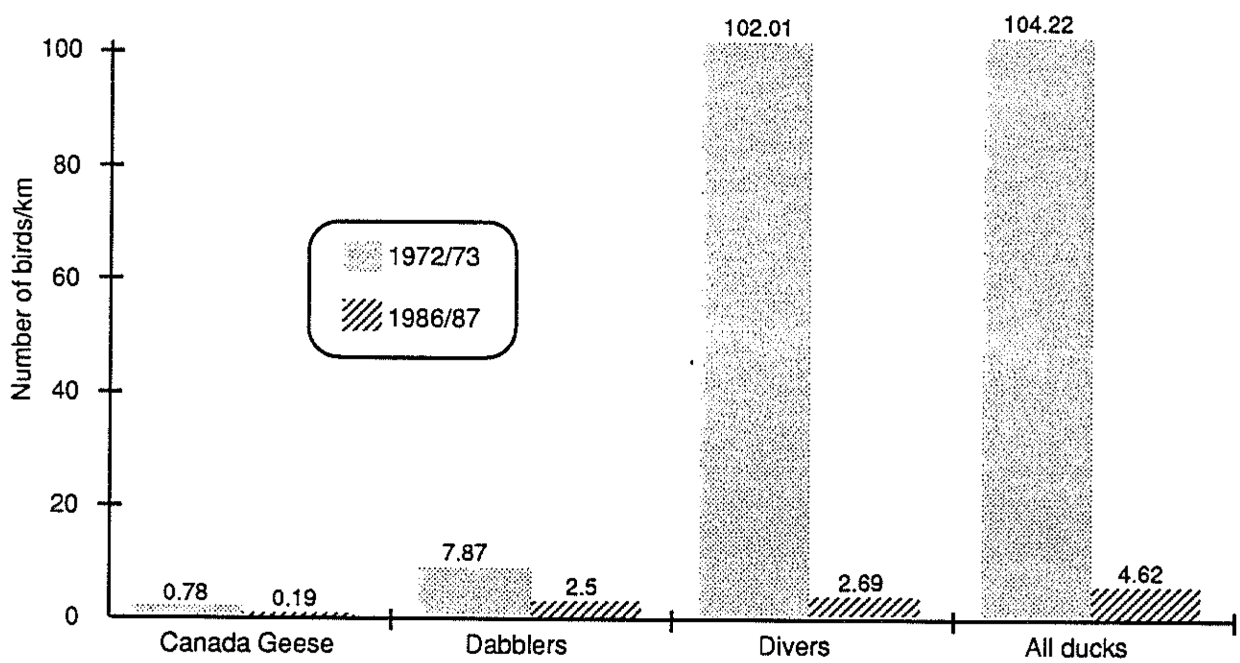

Figure 10. Pre- and post-regulation use of Playgreen–North Basin area by waterfowl based on highest survey densities.

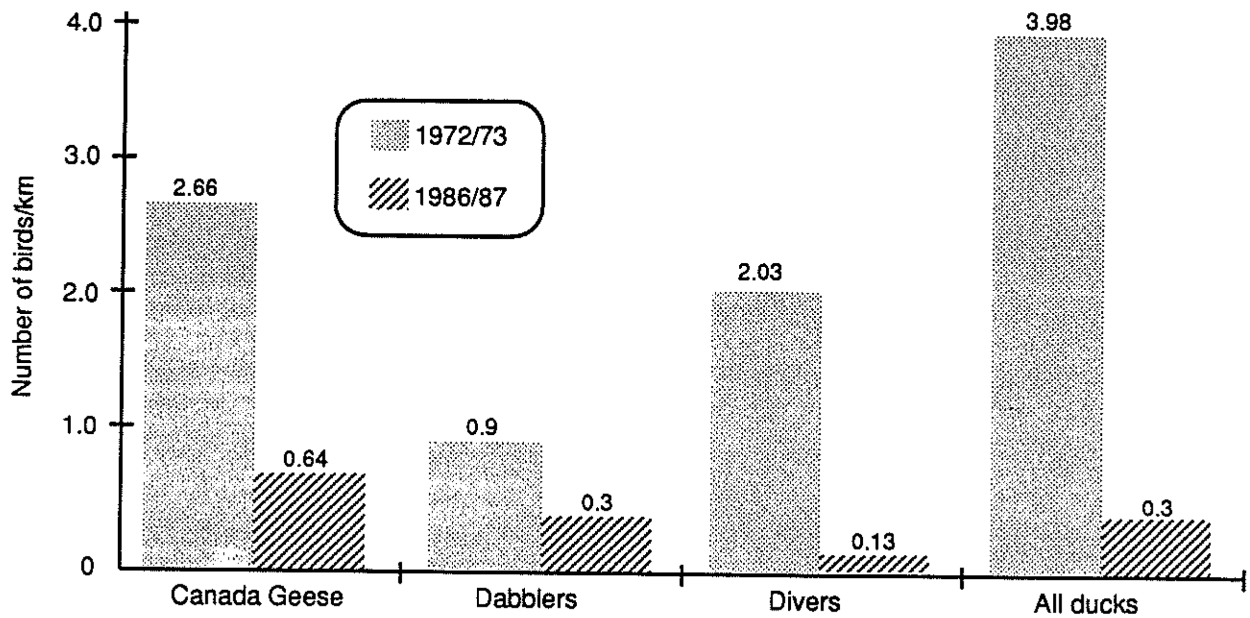

Figure 11. Pre- and post-regulation use of Kiskitto Lake by waterfowl based on highest survey densities.

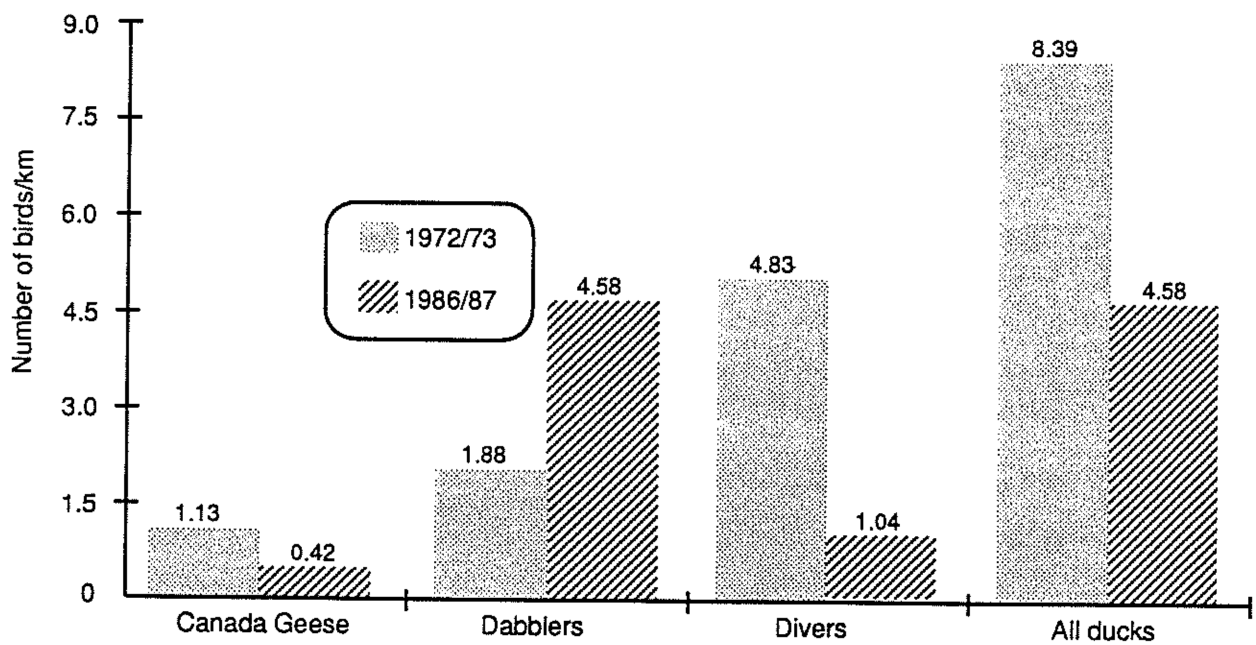

Figure 12. Pre- and post-regulation use of Kiskittogisu Lake by waterfowl based on highest survey densities.

Table 11. Changes in waterfowl densities* in the Outlet Lakes area from pre-regulation to post-regulation based on highest survey densities.

Species by Area	Pre-regulation 1972/73	Post-regulation 1986/87	% Change
PLAYGREEN LAKE — NORTH			
Divers	102.01	2.69	-97.4
All ducks*	104.22	4.62	-95.6
PLAYGREEN LAKE — SOUTH			
Divers	72.07	2.37	-96.7
All ducks*	72.18	4.86	-93.3
KISKITTO LAKE			
Divers	2.03	0.13	-93.6
All ducks*	3.98	0.3	-92.5
KISKITTOGISU LAKE			
Divers	4.83	1.04	-78.5
All ducks**	8.39	4.58	-45.4

* dabbling ducks and Canada geese have been omitted because of gaps and inconsistencies in survey counts
** includes dabblers, divers, & unidentified ducks.

Table 12. Summary of changes in waterfowl use of the Outlet Lakes area from pre-regulation to post-regulation based on highest survey densities.

Species	Pre-diversion 1972/73	Post-diversion 1986/87	% Change
Divers	41.85	1.57	-96.2
All ducks*	43.51	4.31	-90.1

* includes dabblers, divers & unidentified ducks.

7.2 Possible Causes of Waterfowl Use Changes

The following possible causes for the observed changes in diving duck numbers in the Outlet Lakes include:

1) Regional changes in waterfowl populations;
2) Modifications to the physical environment associated with Lake Winnipeg regulation; and
3) Habitat changes associated with Lake Winnipeg regulation.

7.2.1 Regional Waterfowl Population Changes

Regional changes in waterfowl populations can result from overexploitation by hunters or as a result of a widespread reduction in the availability or quality of habitat. If waterfowl populations had declined throughout northern Manitoba, one would expect such declines to be represented in waterfowl numbers in more localized areas such as the Outlet Lakes.

To determine whether the observed reduction in waterfowl use of the Outlet Lakes area could be attributed to a more widespread, regional decline in waterfowl populations, data collected annually by the U.S. Fish and Wildlife Service during the breeding pair surveys in Stratum 24 in northern Manitoba (Figure 13), were examined. These data provide adjusted population estimates for ducks and indicated breeding pair numbers for Canada geese by stratum and serve as general indicators of population size and trends.

It is important to remember that these data estimate the size of breeding populations of ducks and geese and do not indicate the size of fall migrating flocks or the "fall flight". However, the data are still useful as broad indices of waterfowl population size and trends.

Data for 1964-75 were selected to describe regional waterfowl population levels before LWR was fully operational, while data for 1977-88 represent regional population levels following regulation. The estimates of dabbler and Canada goose numbers are provided for general reference and indicate that, while

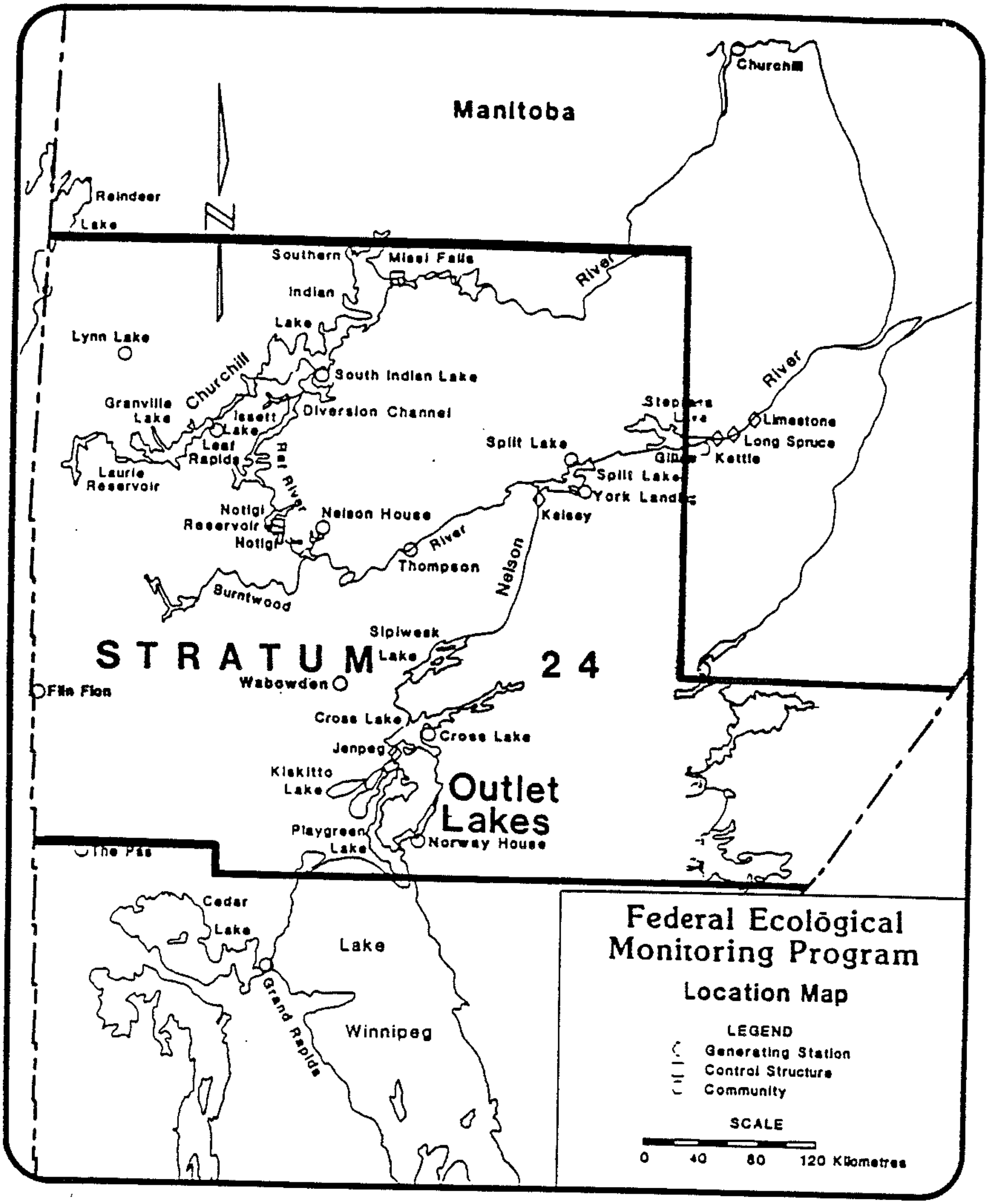

Figure 13. Location of waterfowl survey Stratum 24.

populations have fluctuated considerably over the past 24 years, there is little evidence of either a sustained increase or decline in over time (Table 13). Diving duck population data for these two periods are also given in Table 13 and are discussed in greater detail below. Regional changes in diver populations that have occurred from 1955 to 1988 are displayed in Figure 14.

Figure 14 shows that large year-to-year fluctuations occurred in regional diving duck populations occurred during both the pre-and post-regulation periods. However, when the 12-year means for the two periods are compared, only a small change in numbers is evident from the pre-regulation period to the post-regulation period for divers (Table 14). Overall, the diver population estimate increased by 2.6 percent. Results of the 1973 and 1988 surveys for the Outlet Lakes area are included for comparison.

One must keep in mind, however, that the means on which these changes are based will vary somewhat as individual year's counts are added or subtracted. The main point to be made is that numbers of breeding waterfowl in the study region have neither declined nor increased over the long term, according to Stratum counts.

In conclusion, there is no indication that the observed reduction in waterfowl use in the Outlet Lakes area reflects a long-term decline in regional populations as represented by the Stratum 24 counts, since no such trends are evident in the stratum data. At the same time, one must be cautious about interpreting the significance of the 96 percent change in diver numbers derived from 1973 and 1988 surveys, because large between-year changes are not uncommon in waterfowl counts derived from aerial surveys.

For example, comparison of any pre-diversion year with any post-diversion year estimate using the Stratum 24 data can yield dramatically different results that may or may not have anything to do with changes in regional water conditions: comparison 1986 and 1970 Stratum counts suggest a 78 percent reduction in dabbling ducks, whereas comparison of 1978 and 1966 counts indicate a 437 percent increase. In the absence of counts compiled over several

Table 13. Adjusted population estimates (thousands) for dabblers, divers, and Canada geese in stratum 24 during pre-regulation (1964-75) and post-regulation (1977-88) periods (from U.S. Fish & Wildlife Service).

PRE-DIVERSION	1964	1965	1966	1967	1968	1969	1970	1971	1972	1973	1974	1975	MEAN
dabblers	228.9	404.4	205.0	716.0	1122.9	1102.3	1458.7	631.2	440.2	500.3	345.1	358.5	626.1
divers	198.2	276.7	228.3	233.0	587.7	653.3	788.0	447.4	513.8	375.5	270.2	383.2	412.9
Totals	**427.1**	**681.1**	**433.3**	**949.0**	**1710.6**	**1755.6**	**2246.7**	**1078.6**	**954.0**	**875.8**	**615.3**	**741.7**	**1039.1**
Canada geese	4560	6952	5145	5050	9504	15606	5925	4700	2498	1845	1590	2041.0	5451.3

POST-DIVERSION	1977	1978	1979	1980	1981	1982	1983	1983	1985	1986	1987	1988	MEAN
dabblers	708.9	1101.5	935.5	799.8	837.2	561.6	584.4	347.2	372.6	248.6	441.3	588.6	627.3
divers	536.2	376.2	336.9	387.6	746.3	329.4	531.7	319.4	359.1	534.1	339.8	288.2	423.7
Totals	**1245.1**	**1477.7**	**1272.4**	**1187.4**	**1583.5**	**891.0**	**1116.1**	**666.6**	**731.7**	**782.7**	**781.1**	**876.8**	**1051.0**
Canada geese	4444	6750	9743	5125	5681	2840	5000	2525	5517	1914	5345	8104.0	5249.0

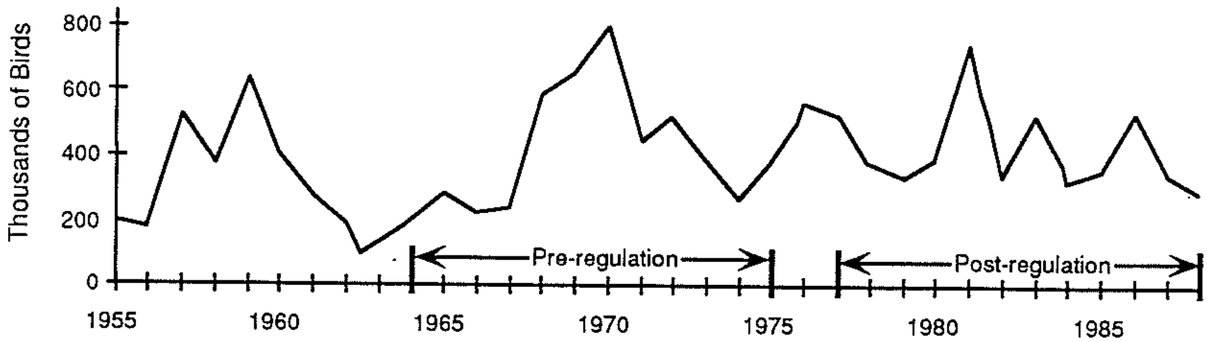

Figure 14. Adjusted population estimates of diving ducks in stratum 24 (northern Manitoba), 1955-1988.

Table 14. Changes in regional (Stratum 24) and local (Outlet Lakes) waterfowl use from pre-regulation to post-regulation.

Waterfowl	Stratum 24: 12-year Means*			Rat-Burntwood Counts**		
	1965-75 (pre)	1978-88 (post)	% Change	1973 (pre)	1988 (post)	% Change
Divers	412.9	423.7	2.6	41.85	1.57	-96.2

* adjusted population estimates (thousands)
** numbers represent birds/km

years, one would be hard-pressed to draw meaningful conclusions from these point-in-time data.

7.2.2 Transmission Lines and Channels

Norway House residents suggested that the transmission line constructed to provide electrical service to Norway House may have caused ducks to alter their flight patterns during fall migration. The transmission line spans both 2-Mile Channel and the natural outlet of Lake Winnipeg near Warren Landing. In addition, 2-Mile Channel itself represents an alteration to the physical environment.

As stated in Section 7.3, hunters from Norway House observed migrating flocks of ducks turn towards Lake Winnipeg at 2-Mile Channel as they crossed the south basin of Playgreen Lake in the fall, rather than using their traditional route past Warren Landing. Since the transmission line crosses both routes, it would appear that this change in flight pattern is more likely to be a result of new navigational cues associated with the new channel, and the shorter access to Lake Winnipeg, rather than avoidance of the transmission line in the Warren Landing area.

According to Manitoba Hydro records (D. Windsor, pers. comm.), the main transmission line to Norway House was in place by the end of the 1972-73 winter construction season (a branch line to Warren Landing was installed during the winter of 1973-74). Consequently, the line was already in place when four of the five pre-regulation aerial waterfowl surveys were conducted in the fall of 1973 (the fifth survey was conducted in the fall of 1972). If the presence of the transmission line caused ducks to avoid the Warren Landing area, one would expect that this effect would have already occurred in the fall of 1973 and would be reflected in the 1973 survey results. It is unlikely, then, that the observed reduction in duck numbers is related to presence of the transmission line.

If the presence of the transmission line and/or 2-Mile Channel was influencing waterfowl use or flight patterns, the effect would likely be localized.

The observed large reduction in diving duck use of the north basin of Playgreen Lake (Figure 11) cannot be attributed to localized changes in physical environmental conditions, such as 2-Mile Channel, which may be affecting local flight patterns.

In conclusion, it is unlikely that waterfowl use of the Outlet Lakes area has been affected directly by the presence of the transmission line or 2-Mile Channel. Consequently, it would not be valid to attribute the decline in waterfowl use to the physical presence of these features.

7.2.3 Habitat Changes

It is possible that the altered water regime, associated with Lake Winnipeg regulation, may have resulted in deterioration of waterfowl feeding and staging habitat and, consequently, reduced the attractiveness of the Outlet Lakes area to waterfowl during the fall staging and migration periods. It is also possible that 2-Mile Channel may have altered the quality of waterfowl habitat.

Increased water levels in Playgreen Lake levels apparently have caused flooding of shoreline vegetation and soils, resulting in increased amounts of sediment entering the Lake and remaining in the water column. MacLaren PlanSearch Inc. (1985) documented the susceptibility of sections of Playgreen Lake south basin shoreline to erosion and referred to the remnants of an old hydro pole situated in the water about 6 m from the southern shoreline in 1984. According to Manitoba Hydro records, the pole had been installed on land in 1972, some 15 to 30 m inland from shore. Based on this information, MacLaren PlanSearch estimated that between 21 and 36 metres of shoreline had eroded during the 12-year period between 1972 and 1984. Increases in turbidity, total organic carbon and total inorganic carbon, have been measured in the Norway House area following LWR and may be attributed to shoreline erosion or construction of 2-Mile Channel (Playle and Williamson 1986).

Water level regimes before and after Lake Winnipeg regulation in Playgreen Lake were examined using the data collected at Water Survey of Canada's Nelson River at Norway House station (#05UB001). The historic record (1915-87) was divided into pre-regulation (1915-74) and post-regulation (1977-87) periods. Data for 1975 and 1976 were excluded because they were considered to be transitional between pre- and post-regulation conditions. Excavation of 8-Mile Channel was not finished until September 13, 1975, with dredging completed on October 6, 1975. Similarly, 2-Mile Channel did not connect Lake Winnipeg to Playgreen Lake until May 12, 1976 and dredging was not completed until October 5, 1976.

Analysis of mean monthly water levels at Playgreen Lake during the pre-regulation and post-regulation periods suggest that mean post-regulation water levels were 8 to 23 cm higher than mean pre-regulation levels (Table 15), with the greatest increase occurring in September. Because the mean monthly values from both pre- and post-regulation periods have not been adjusted for variance within or between months and years, the significance of the monthly differences shown in Table 15 cannot not determined. For example, the range of variation reported from daily records for August, September, and October 1986 was 23, 53, and 41 cm, respectively, at Playgreen Lake. Similarly, comparison of mean monthly water levels from two sample years before regulation (1972/73) ranged from 2 to 67 cm; mean monthly differences for two post-regulation years (1986/87) ranged from 1 to 34 cm.

Comparison of pre- and post-regulation figures do indicate higher water levels prevailed during all months following regulation, which would be expected to alter shoreline conditions, including aquatic and near-shore terrestrial habitats. Therefore, it is reasonable to hypothesize that the observed changes in diving duck use of the Outlet Lakes area is associated with habitat changes resulting from the altered flow regime (i.e., slightly increased water levels on a year-round basis) following Lake Winnipeg Regulation.

That increased water levels and associated habitat change can affect use by diving ducks has been documented by Paloumpis and Starrett (1960) and Mills *et al.* (1966) along Illinois River.

Aside from the obvious ecological differences between aquatic systems in Illinois and northern Manitoba, the Illinois River and LWR projects share certain key characteristics. In both, the natural flow regime has been modified by man's activities, and in both cases, diving ducks have experienced the largest reduction in numbers.

Researchers have linked the tremendous decline in the number of lesser scaup using the Illinois River in the fall to the disappearance of fingernail clams (Sphaeridae) and small snails (Gastropoda) which were the main food items of lesser scaup at that location (Anders 1959, Rogers and Korschgen 1966). Increased turbidity and siltation and a substantial reduction in the abundance of aquatic plants have also been documented through various studies (Mills et al. 1966).

Table 15. Pre- and post-regulation water levels (m asl) at Playgreen Lake-Norway House hydrometric station (#05UB001).

	JAN	FEB	MAR	APR	MAY	JUN	JUL	AUG	SEP	OCT	NOV	DEC
LONG-TERM MEANS												
1978-88 (Post)	217.04	217.03	217.04	217.14	217.29	217.47	217.49	217.49	217.38	217.27	217.12	217.02
1969-75 (Pre)	216.92	216.89	216.88	216.97	217.10	217.29	217.33	217.30	217.15	217.09	217.02	216.94
Difference*	0.12	0.11	0.16	0.17	0.19	0.18	0.16	0.19	0.23	0.18	0.10	0.08

* Given the degree of variation in water levels recorded both within months and between years, pre- and post-regulation differences shown here may not be statistically significant.

The fact that dramatic reductions in diving duck use accompanied alterations to the natural flow of water in both areas prompted a more detailed evaluation of waterfowl response to changes in water regime. Findings are presented in the following section.

7.3 Assessment of Habitat Change Impacts on Waterfowl Use

7.3.1 Methodology

To determine how the altered flow regime could affect waterfowl use of the Outlet Lakes area, a Valued Ecosystem Components (VECs) approach (Beanlands and Duinker 1983) was used to analyze those components of the environment having particular relevance to the specific assessment being conducted. The approach was used successfully in evaluating the potential environmental impacts of hydrocarbon development in the Beaufort Sea through the Beaufort Environmental Monitoring Project (Indian and Northern Affairs Canada 1983). More recently, the VEC concept was used to develop a screening system for identifying potential environmental concerns associated with Public Work Canada dredging proposals (Vonk et al. 1989).

The approach consisted of the following steps:

a) identification of the VECs;
b) identification of impact hypotheses that relate changes to the physical environment to the VECs; and
c) evaluation of impact hypotheses to determine validity.

7.3.2 Valued Ecosystem Components

First, the scientific literature dealing with the feeding habits of ducks; the relationships between benthic invertebrates and aquatic macrophytes; and the effects of turbidity, siltation and water level changes on macrophytes and invertebrates was thoroughly reviewed and the following VECs identified:

a) one or more diving duck species present;

b) one or more benthic invertebrate groups (e.g. amphipods) present;

c) one or more submerged macrophyte species (e.g. pondweeds) present; and

d) one or more components representing habitat attractiveness to ducks.

Lesser scaup was selected as a VEC because it is one of the most common diving duck species found in the Outlet Lakes area, and its food habits are well documented in the literature.

Table 16 summarizes the results of lesser scaup food selection studies. The preference towards animal foods represented by benthic invertebrates is immediately obvious. Of the invertebrates consumed, Gastropoda (snails), Pelecypoda (particularly Sphaeriidae, the fingernail clams), Amphipoda (amphipods) and Trichoptera (particularly caddis fly larvae) emerge as important food items. Due to the variety of ecological niches occupied by these different invertebrate types, "benthic invertebrates" were identified collectively as being a VEC, with each of the four groups representing "sub-VECs".

The existence of pondweed beds in Playgreen Lake is well known. MacLaren Plansearch Inc. (1985) described the distribution and abundance of *Potomogeton vaginatus* and *P. richardsonii* along the southern shoreline of the south basin of Playgreen Lake. The authors observed that *P. richardsonii typically* grows in shallower littoral zones, while *P. vaginatus* tends to inhabit deeper water. They also noted that the production of *P. richardsonii* may have diminished as a result of increased Lake levels associated with Lake Winnipeg regulation. For this reason, "pondweeds" were identified collectively as a VEC, with *P. richardsonii* and *P. vaginatus* as sub-VECs.

The means by which waterfowl navigate, recognize geographical landmarks and select habitats to meet their needs is not well-understood (e.g. Bellrose 1972, Wiltschko and Wiltschko 1978). This lack of understanding is not limited to waterfowl but, applies to all birds. Notwithstanding this knowledge deficiency, it is evident that waterfowl and other bird species do possess the ability to

Table 16. Food items (by % of diet) consumed by lesser scaup in Alberta, Saskatchewan, Manitoba, Northwest Territories, Minnesota, and Illinois.

Food Items	Sugden (1973), Alberta *	Dirschl (1969), Saskatchewan**	Bartonek & Hickey (1969), Manitoba^	Bartonek & Murdy (1969), N.W.T*	Afton et al. (press) Minnesota**	Anderson (1959) Illinois **	Rogers & Korschgen (1966), Illinois**
PLANT MATERIAL							
Carex spp.		0.4					
Lemna		0.5			0.3		
Myriophyllum spp.	2.0	0.3	5.0				
Scirpus spp.		2.2	7.0		0.2		2.9
Potamogeton spp.		3.3			0.1	2.8	3.3
Ceratophyllum demersum						2.4	
Misc. plants	2.0		2.0	1.0	6.6	4.5	0.3
Total Plants	**4.0**	6.7	**14.0**	**1.0**	**7.2**	**9.7**	**6.5**
ANIMAL MATERIAL							
Mollusca							
Gastropoda	16.0		3.0	1.0	3.0	43.4	70.1
Pelecypoda							
Sphaeriidae						32.3	11.9
Unionidae						2.6	2.9
Unidentified						0.3	0.1
Misc. Mollusca						7.6	
Total Mollusca	**16.0**	**0.0**	**3.0**	**1.0**	**3.0**	**86.2**	**85.0**
Arthropoda							
Crustacea							
Amphipoda	52.0	85.0	7.0	57.0	74.5		
Misc. Crustacea	1.0			2.0	2.9		0.5
Total Crustacea	**53.0**	**85.0**	**7.0**	**59.0**	**77.4**	**0.0**	**0.5**
Insecta							
Ephemeroptera						3.3	7.8
Odonata	3.0			17.0	0.2	0.2	
Hemioptera	3.0			11.0		0.3	
Coleoptera	1.0		2.0	4.0		0.1	
Diptera	16.0	0.3	12.0	1.0	8.9	0.1	
Trichoptera	2.0	4.5	24.0	6.0	1.9		
Misc. Insecta	1.0		1.0		0.2		0.2
Total Insecta	**26.0**	**4.8**	**39.0**	**39.0**	**11.2**	**4.0**	**8.0**
Hirudinae	1.0	3.5	31.0		1.2		
Misc. Animal			6.0				
Total Animals	**96.0**	**93.3**	**86.0**	**99.0**	**92.8**	**90.2**	**93.5**

**fall, both sexes; ^summer, males only; *late summer, both sexes recognize habitat types capable of fulfilling their needs and to distinguish good-quality habitat from poor-quality or unsuitable habitat (Hilden 1965). For this reason, a VEC representing a set of visual cues, related to landscape or habitat appearance, was identified. This VEC was termed "landscape characteristics".

In summary, four VECs were identified to represent "cause-and-effect" hypotheses:

a) lesser scaup;
b) benthic invertebrates;
c) pondweeds; and
d) landscape characteristics.

Four sub-VECs under "benthic invertebrates" are:

i) gastropods;
ii) amphipods;
iii) fingernail clams; and
iv) caddisfly larvae.

There are two sub-VECs under "pondweeds":

i) *P. richardsonii*; and
ii) *P. vaginatus*.

No sub-VECs were identified for the VEC "landscape" characteristics.

7.3.3 Impact Hypotheses

Figure 15 illustrates the possible linkages between increased water levels and reduced waterfowl use (diving duck use) of Playgreen Lake. Each linkage has been numbered for ease of reference.

An "impact hypothesis" can be defined as a statement that links a development-related activity to an environmental effect. In this case, the development-related activity is an increase in water levels, related to regulation of Lake Winnipeg, and the environmental effect is reduced diving duck use. Examination of the network diagram in Figure 15 indicates that two distinct "impact hypotheses" can be constructed to explain the relationship between water levels and waterfowl (diving duck) use:

A. **Increased water levels, resulting from regulation of Lake Winnipeg, have caused a reduction in the availability of invertebrate food items and, consequently, a reduction in the attractiveness of Playgreen Lake to diving ducks.**

B. **Increased water levels, resulting from regulation of Lake Winnipeg, have modified the landscape and habitat characteristics and, consequently, have reduced the attractiveness of Playgreen Lake to diving ducks.**

7.3.3.1 Hypothesis A

To determine whether an increase in the level of Playgreen Lake could have resulted in a reduction in the availability of invertebrates as food for diving ducks, it is necessary to investigate some of the factors which control or influence the abundance of invertebrates in aquatic ecosystems.

Potamogeton richardsonii favours the shallower littoral zones of Playgreen Lake while *P. vaginatus* tends to inhabit the zones of deeper water. MacLaren Plansearch Inc. (1985) acknowledged that *P. richardsonii* may have diminished as a result of increased lake levels associated with Lake Winnipeg regulation. In fact, it is likely that increased lake levels have caused stands of *P. vaginatus* to become more widespread at the expense of *P. richardsonii.* Robel (1962) documented the sensitivity of *P. pectinatus* to water level changes. A rise in water level of less than 8 cm caused vegetation production to increase by 32 percent in shallow areas and decrease by 35 percent in the deeper areas.

The association of high densities and diversity of invertebrates with aquatic macrophytes, including pondweeds such as *P. vaginatus* and *P. richardsonii*, is well documented in the literature. Researchers have noted that benthic invertebrates inhabiting lake substrates, such as snails and fingernail clams, are much more abundant in vegetated areas than in non-vegetated areas (e.g. Krull 1970, Engel 1988). Voigts (1976) found that amphipods were most abundant in dense beds of submerged vegetation.

Many benthic invertebrate species depend on aquatic macrophytes to meet essential life cycle requirements. The leaves of aquatic plants provide a substrate for egg deposition by several invertebrate species, including snails (Gerrish and Bristow 1979, Pandit 1984). Macrophytes support large quantities of periphyton (a film of microorganisms, chiefly bacteria, protozoa and algae), a favoured food source for amphipods (Brown et al. 1988) and snails (Pandit 1984). Caddisfly larvae use the leaves of submerged macrophytes for building dwelling cases (Soszka 1975). Biggs and Malthus (1982) noted small bivalve mussels, family Sphaeriidae, were associated with the roots of all macrophytes.

Some submerged aquatic plants provide habitat for a greater diversity and number of invertebrates than others. Gerrish and Bristow (1979) concluded that growth form and leaf morphology of aquatic plants are the main factors determining the size of the resident macroinvertebrate population. Dvorak (1987) noted that, despite their lower biomass, submerged plants offer much larger surface area for macroinvertebrate colonization than emergent plants, which have a much smaller surface area in relation to their higher biomass.

The importance of surface area of submerged aquatic plants in determining the number and diversity of invertebrates present was noted by Krecker (1939), who observed that the pondweed *P. crispus*, which has crenulate leaves, supported more than twice as many invertebrate genera and numbers as *P. compressus* and *P. pectinatus*, which have narrow, thread-like leaves.

Berg (1949) studied insects and insect parasites associated with 17 species of *Potamogeton*. For the Outlet Lakes study, the 17 species were classified into broad-leaved and narrow-leaved group. Table 17 shows that more insect species

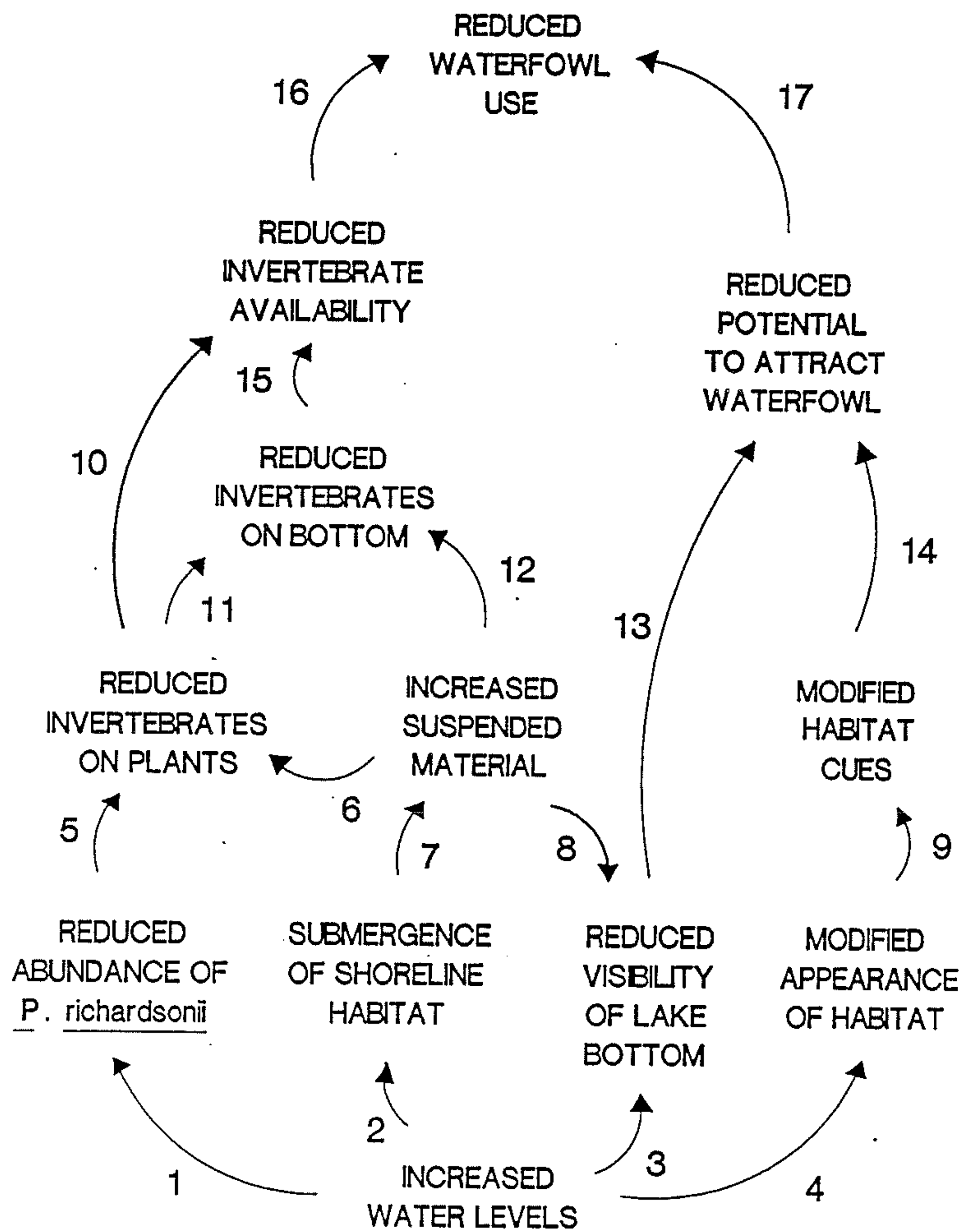

Figure 15. Network diagram showing relationship between waterfowl use and various environmental factors in the Playgreen Lake area.

were found on broad-leaved species than on narrow-leaved *Potamogeton*. Berg pointed out that caddisfly larvae can obtain more abundant and better materials for casemaking from broad-leaved aquatic plants than from narrow-leaved plants.

It was noted earlier that increased lake levels would cause stands of *P. richardsonii* to decline (linkage 1 in Figure 15) and *P. varnatus* to increase. Since *P. richardsonii* is a broad-leaved pondweed, while *P. vaginatus* is narrow-leaved, a reduction in the distribution and abundance of *P. richardsonii* would lead to an overall reduction in the abundance of invertebrates (linkage 5). Several aquatic invertebrate groups (e.g. amphipods, snails) spend part of their lives on the submerged plant substrate, but also inhabit the bottom substrate where plants are growing. Therefore, a reduction in invertebrate numbers on plants could be accompanied by a reduction in invertebrate numbers on the bottom substrate (linkage 11), although the existence of such a cause-effect relationship would have to be verified through field studies. The combined effect would be a reduction in the availability of invertebrates as food for diving ducks (linkages 10 and 15). With less animal food available, Playgreen Lake would be less attractive to diving ducks and waterfowl use would be expected to decline (linkage 16).

Flooding and submergence of shoreline vegetation and soils (linkage 2) due to increased water levels is expected to increase the amount of sediment entering Playgreen Lake and remaining in the water column (linkage 7). It should be noted that the 0.75 to 0.78 m increase in water depth at Playgreen Lake is lower than historic high water levels recorded before LWR. Nevertheless, water levels will remain high for long periods under the regulated regime; this, coupled with the reversal of the pre-regulation flood-drawdown cycle, has likely affected shoreline erosion patterns and growth of aquatic vegetation.

Increased levels of sediment adversely affect aquatic invertebrates but benefit others, both directly (linkage 12) and indirectly through effects on aquatic macrophytes (linkage 6).

Ellis (1936) observed that most of the common freshwater mussels (Mollusca) were unable to maintain themselves on sand or gravel bottoms (normally satisfactory bottom habitat) that were blanketed with silt, even when silt

Table 17. Numbers of insect species observed on broad-leaved and narrow-leaved pondweeds (Berg 1949).

Pondweed Species	No. of Insect Species (including parasites)	No. of Insect Species (no parasites)
BROAD-LEAVED		
P. alpinus	13	8
P. amplifolius	25	20
P. gramineus	10	8
P. illinoensis	5	5
P. natans	23	16
P. nodosus	6	5
P. oakesianus	4	1
P. praelongus	10	7
P. richardsonii	18	16
MEAN	**12.7**	**9.6**
NARROW-LEAVED		
P. epihydrus	6	4
P. filiformis	0	0
P. foliosus	2	2
P. friesii	1	0
P. pectinatus	1	1
p. pusillus	0	0
P. robbinsii	6	6
P. zosteriformis	7	4
MEAN	**2.9**	**2.1**

deposition was less than 3 cm deep. High mussel mortality was found to be induced by the silt covering rather than low oxygen, pH, or other water quality characteristics. Laboratory experiments showed that silt interfered with the feeding of freshwater mussels.

Siltation, or sedimentation, is believed to have been a major factor in the almost complete disappearance of pondweeds and other large aquatic plants along the Illinois River in Illinois (Mills et al. 1966). The authors state that "siltation affects aquatic plants adversely in two ways: it produces a turbidity which reduces the penetration of light and inhibits photosynthesis, and it creates bottom conditions which make it difficult or impossible for various species of plants to obtain anchorage when they are buffeted by wave action". The great reduction in aquatic plants was accompanied by virtual disappearance of snails and fingernail clams. Siltation was considered to be an important factor influencing the drastic reduction in bottom fauna (Paloumpis and Starrett 1960).

Paloumpis and Starrett (1960) attributed the scarcity of lesser scaup at Quiver Lake, on the Illinois River, to the disappearance of small snails and the decline of fingernail clams. Mills et al. (1966) similarly linked the major decline in lesser scaup along Illinois River in fall to disappearance of fingernail clams, citing Anderson's (1959) findings that mollusca comprised over 85 percent of the diet of lesser scaup (Table 16).

Increased sedimentation and turbidity, combined with increased water depth resulting from a rise in Lake level, would cause a reduction in visibility and could interfere with the ability of diving ducks to locate food items in the water column and on the bottom of Playgreen Lake (linkages 3 and 8).

In a controlled-environment study using a large aquarium, Tome and Wrubleski (1988) noted that lesser scaup appeared to locate prey visually. The authors observed that diving scaup frequently stopped and directed foraging movements of the bill towards amphipods swimming in the water column. They consumed prey distributed on artificial vegetation placed in the aquarium and often pursued and consumed amphipods that would drop to the substrate when the 'vegetation' was disturbed. Increased turbidity combined with increased depth would tend to reduce visibility, and would likely make such waters less attractive to diving ducks.

Results of surveys conducted by Fisheries and Oceans Canada (DFO) in 1987 and 1989 indicated that standing stocks of various benthic invertebrate groups inhabited the south and north basins of Playgreen Lake and in Kiskittogisu Lake (McKerness 1990). However, these results do not necessarily mean that those invertebrates were available to diving ducks such as scaup. Benthic invertebrate sampling stations were located on open water transects which did not include the near-shore vegetated zones of the lakes sampled (D. Rosenberg, pers. comm.). Therefore, while the margins of the diving duck feeding areas were not covered by the DFO surveys, open-water areas typically used by diving ducks for foraging were sampled. In addition, the DFO surveys were both conducted in July. No data was collected on the abundance of benthic invertebrates during the fall waterfowl staging period of September and October. Consequently, no firm conclusions can be drawn regarding the abundance or availability of benthic invertebrates to fall-staging diving ducks such as scaup.

The above analysis suggests Impact Hypothesis A (that increased water levels, associated with Lake Winnipeg regulation, may have reduced the availability of invertebrate food items to diving ducks, and contributed a decline in diving duck use of the Outlet Lakes area) has some validity, although field studies are required for verification.

7.3.3.2 <u>Hypothesis B</u>

Hypothesis B assumes that increased water levels have modified the landscape and habitat characteristics of the Outlet Lakes area, thereby reducing the area's attractiveness for diving ducks.

Although many scientific papers have been published on the topics of bird navigation and habitat selection, the level of understanding of the actual mechanics involved remains poor (Bellrose 1976). Hilden (1965) in his review of habitat selection mechanisms in birds during the breeding season, refers to "proximate" and "ultimate" factors which operate together and result in the selection of habitat.

Ultimate factors are essential for the survival of a species and include: food, requirements imposed by structural and functional characteristics of the species, and shelter from enemies and adverse weather. Proximate factors are certain characteristic stimuli of the species-specific habitat that release what is referred to as the "settling reaction". These factors include: landscape, terrain, feeding and drinking sites and other animals.

Not every suitable habitat need possess all the features characteristic of the optimal environment. Rather, it appears the positive and negative habitat characteristics are "summed" and, when the combined effect of the stimuli exceeds the threshold of the settling reaction, the habitat is selected. Similarly, one key stimulus, such as adverse landscape appearance, may out-weigh the other stimuli; in that situation, the combined effect of other positive stimuli would not be sufficient to induce the bird to "settle" in that particular habitat. Studies on wild geese conducted by Barry (1962) and Wood (1964) concluded that, where certain proximate factors such as appropriate nesting habitat conditions are lacking, ovulation can be inhibited and the nesting cycle terminated. Marshall (1952) speculated that inhibitions to breeding in birds appear to be brought about by the lack of one or more important environmental stimuli necessary to complete the reproductive cycle.

Habitat selection is probably a two-stage process (Hilden 1965). The first stage, settling down and exploring a particular habitat area, is released by features of the landscape and general characteristics of the terrain. Whether the habitat is suitable or not depends on how closely certain of its details conform to the other stimuli constituting the proximate factors in the bird's habitat selection mechanism.

Landscape and terrain are important "proximate" factors influencing habitat selection. Sometimes, the key factor for habitat selection may seem subtle but is still biologically important. For example, the lapwing (*Vanellus vanellus*), prefers grey-brown meadows and avoids green ones. The former are poor meadows where the grass will remain low, whereas the latter are fertile and the grass will later grow high. As the lapwing is adapted to live in low grass, this selection mechanism guides the bird to the most suitable environment. The colour of the meadow (a proximate factor) serves as a reliable indicator of a favourable breeding place (an ultimate factor).

At Playgreen Lake, there are a number of negative environmental stimuli, related to habitat quality, that may serve to repel diving ducks. Higher lake levels have modified the appearance of shoreline and off-shore habitat (linkage 4). The band of emergent vegetation along the shoreline has been decimated by deeper water. Turbidity levels in the Lake have increased which, in turn, has decreased visibility through the water column. In short, higher lake levels have modified (linkage 9) habitat characteristics (cues), thereby reducing the potential of the habitat to attract diving ducks (linkage 14).

While the reaction of birds to stimuli guiding them to suitable habitat is initially inherited, it is apparent that this innate behaviour is either reinforced or modified at a later stage by learning (Hilden 1965). Therefore, while Playgreen Lake used to attract reasonably large numbers of diving ducks, it appears that, at some point following Lake Winnipeg regulation, the ducks may have discovered that the quality of habitat had declined to a point where it did not meet their requirements for staging and, therefore, received less use during fall (linkage 17).

Conditions in the Outlet Lakes area following LWR tend to support the validity of Impact Hypothesis B: that increased water levels, associated with Lake Winnipeg regulation, have reduced the attractiveness of the Outlet Lakes area to diving ducks through modification of landscape and habitat characteristics.

8. SUMMARY AND CONCLUSIONS

8.1 Project Impacts on Waterfowl and Waterfowl Habitat

Comparison of waterfowl survey data collected in 1972/73 and 1986/87 suggests that diving duck use of the Outlet Lakes area in fall has declined following completion of the LWR Project in 1976. Results of pre- and post-regulation surveys were insufficient to estimate changes in levels of use by dabbling ducks and Canada geese with any degree of confidence.

Densities of diving ducks observed along shorelines of Outlet Lakes in the fall of 1986 and 1987 were approximately 96 percent lower than densities recorded prior to Lake Winnipeg regulation in 1972 and 1973. The largest reductions (97 percent) in diving duck use have occurred in the north and south basins of Playgreen Lake.

The observed decline in the use of the Outlet Lakes area by divers cannot be attributed to changes in regional waterfowl populations documented in northern Manitoba. There is no indication from Stratum 24 counts that regional diver populations in the study region have increased or declined in the long term, although their numbers have fluctuated dramatically from year to year.

Analysis of water level data for Playgreen Lake indicates that slight year-round increases in water levels following Lake Winnipeg regulation have altered the quality of Playgreen Lake habitat and may have reduced the attractiveness of Playgreen Lake to diving ducks in the fall.

8.2 Project Impacts on the Norway House Band

Many, if not most, of the preferred duck hunting areas traditionally used by Norway House Bank Indians are located in the south basin of Playgreen Lake, where the greatest reduction in diving duck use was observed. These areas include the eastern shoreline of the Lake south of Norway House, the narrows near Warren Landing, Kettle Island, and Sepastak Channel.

Norway House residents interviewed during the study indicated that reduction in waterfowl availability has limited opportunities for Norway House Band members to hunt and view waterfowl.

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APPENDIX A

NORTHERN FLOOD AGREEMENT

Background

The **Northern Flood Agreement (NFA)** was signed on December 16, 1977 by Canada, Manitoba, Manitoba Hydro and the Northern Flood Committee to establish a mechanism for compensating members of the Nelson House, Norway House, Cross Lake, Split Lake and York Landing (formerly York Factory) bands for adverse effects caused by the hydroelectric development.

The Preamble of the Agreement lays out the premises and principles which together provide a basis for the Agreement's provisions. Clauses C, D, E and G of the Preamble are particularly relevant to the present study and are reproduced below verbatim:

> *C.* *As a result of the modification of the water regime, adverse effects have occurred and may continue to occur, on the lands, pursuits, activities and lifestyles, of the residents, individually and collectively, of the Reserves of Cross Lake, Nelson House, Norway House, Split Lake and York Landing ("the Reserves" as hereinafter defined);*
>
> *D.* *The parties wish to ensure that all persons as defined herein, who may be, or have been, directly or indirectly, adversely affected by the* [Lake Winnipeg Regulation and Churchill River Diversion] *Project shall be dealt with fairly and equitably;*
>
> *E.* *Uncertainty as to the effects of the Project, with respect not only to the Project as it exists at the date of this Agreement but also as it may develop in he future, is such that it is not possible to foresee all the adverse results of the Project nor to determine all those persons who may be affected by it, and, therefore it is desirable to establish through the offices of a single arbitrator a continuing arbitration instrument, to which any person may submit a claim,*

and as well as to fully empower such arbitrator to fashion a just and appropriate remedy;

G. *Canada, by virtue of its jurisdiction and responsibility for Indians and land reserved for Indians, is committed to playing an active role in providing opportunity for the continued viability of the communities and, in particular but without limitation, in making available resources and expertise to the communities in planning and improving the social and economic conditions of the communities, and in ensuring that the special rights of Indians, including those arising from Treaty 5, are adequately protected.*

Contractual Obligations of the Northern Flood Agreement

The provisions of the **NFA** are arranged in a set of 23 Articles which represent contractual obligations. Three subarticles ultimately lead to the initiation of the present study and are, therefore, reproduced below verbatim:

17.1 *Hydro and Canada and Manitoba, severally and jointly, undertake to implement such recommendations of the Lake Winnipeg, Churchill and Nelson Rivers Study Board Report which affect the communities and which fall within their respective or joint jurisdictions.*

17.5 *In particular but without limitation, monitoring of adverse effects of the Project pursuant to the Lake Winnipeg, Churchill and Nelson Rivers Study Board recommendations shall be planned and implemented so as to provide information as may be necessary to give effect to this Agreement.*

18.2 *Canada and Manitoba recognize that the Project is intended to benefit all citizens of Canada, and most particularly of Manitoba, on the one hand, and that the resource users have been and may continue to be adversely affected on the other hand, and that it is in the public interest to ensure that any damage to the interests, opportunities, lifestyles and assets of those adversely affected be compensated appropriately and justly.*

Claim 18

In December 1981, the five NFA Indian Bands and the Northern Flood Committee filed a claim (Claim 18) alleging that Canada, Manitoba and Manitoba Hydro (the Respondents) had failed to implement recommendations of the Lake Winnipeg, Churchill and Nelson Rivers Study Board and to report to the NFA bands on implementation measures.

In particular, they claimed that the Respondents had "failed to establish a mechanism to deal with monitoring and analysis of ongoing social and economic changes related to hydro-electric development and more generally, northern development pursuant to Recommendation 5(c) of the Study Board". In addition, they claimed that the Respondents had "failed to develop and implement a long-term coordinated ecological monitoring and research program to allow impact evaluation and to assist in the ongoing management of the area affected by the Hydro Project pursuant to Recommendation 10 of the Study Board".

Failure to implement these two recommendations meant that the adverse ecological effects of the Project, and the associated adverse social and economic effects, had not been monitored and the information necessary to give effect to the NFA not provided. Consequently, the requirements of NFA sub-article 17.5 had not been met. Furthermore, without this information, it has not been possible to determine to what extent damages to the interests, opportunities, lifestyles and assets of resource users have occurred and, therefore, the level of compensation that is necessary (sub-article 18.2).

Filing of the claim resulted ultimately in approval of funds by Treasury Board on February 6, 1986, for implementation of the NFA over the period 1986-91 (T.B. Minute 800953). Availability of these funds permitted the Canadian Wildlife Service to initiate studies of the adverse effects of the project on waterfowl use of the Rat-Burntwood rivers system in the fall of 1988.