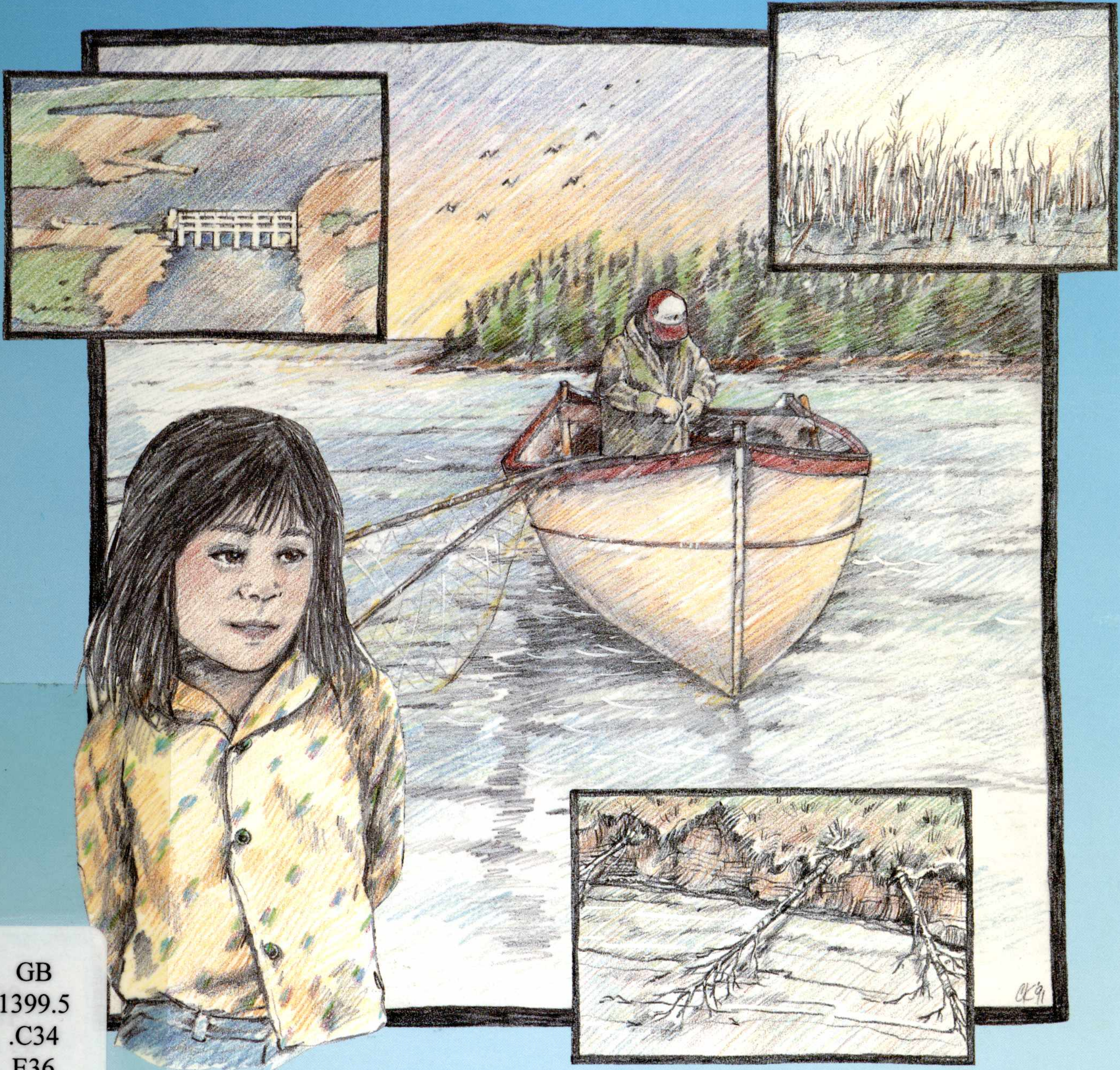
FEDERAL ECOLOGICAL MONITORING PROGRAM

Final Report

Volume 2



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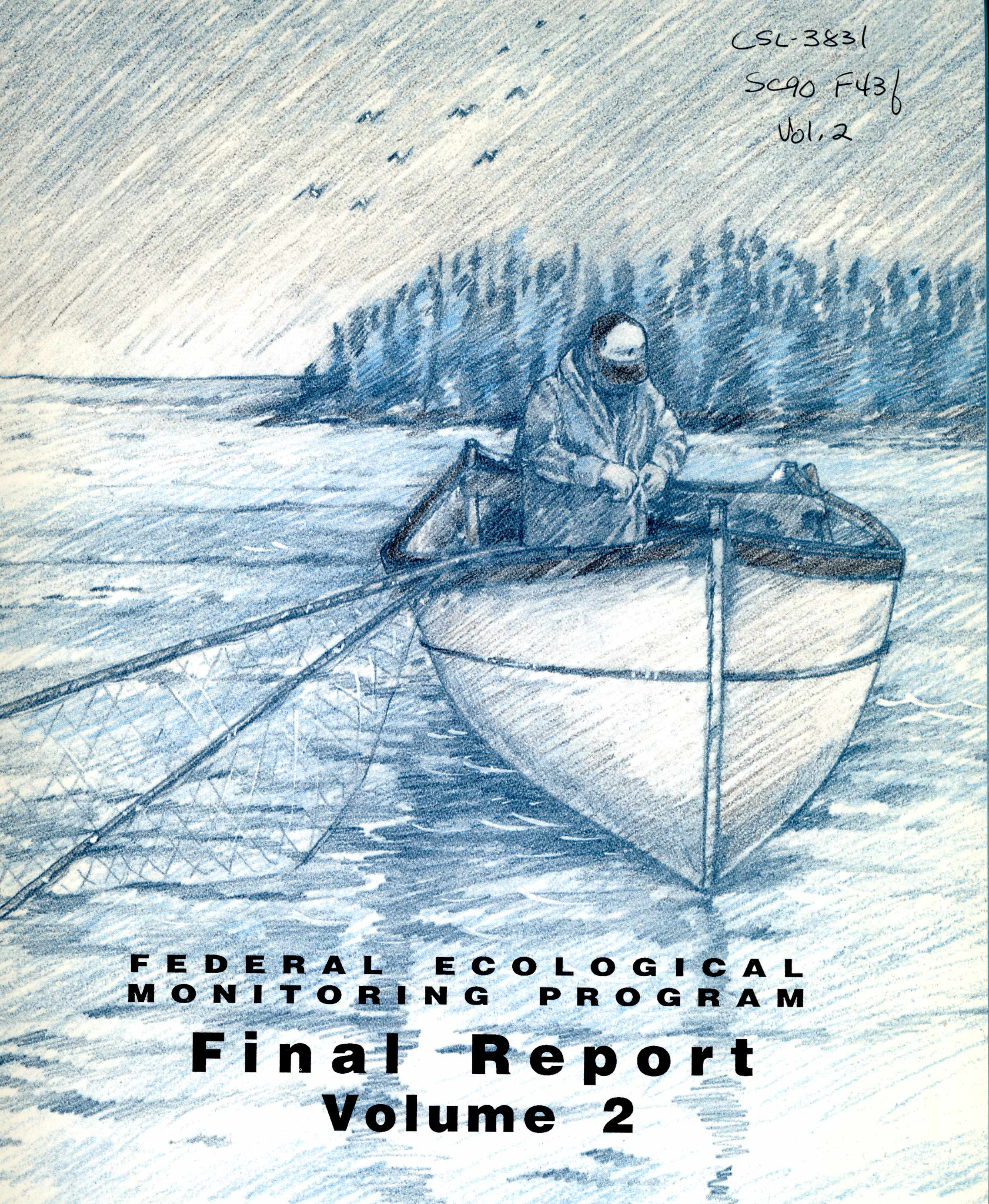

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Final Report

Volume 2

Environment Canada
Conservation and Protection
Western and Northern Region

Department of Fisheries and Oceans
Central and Arctic Region
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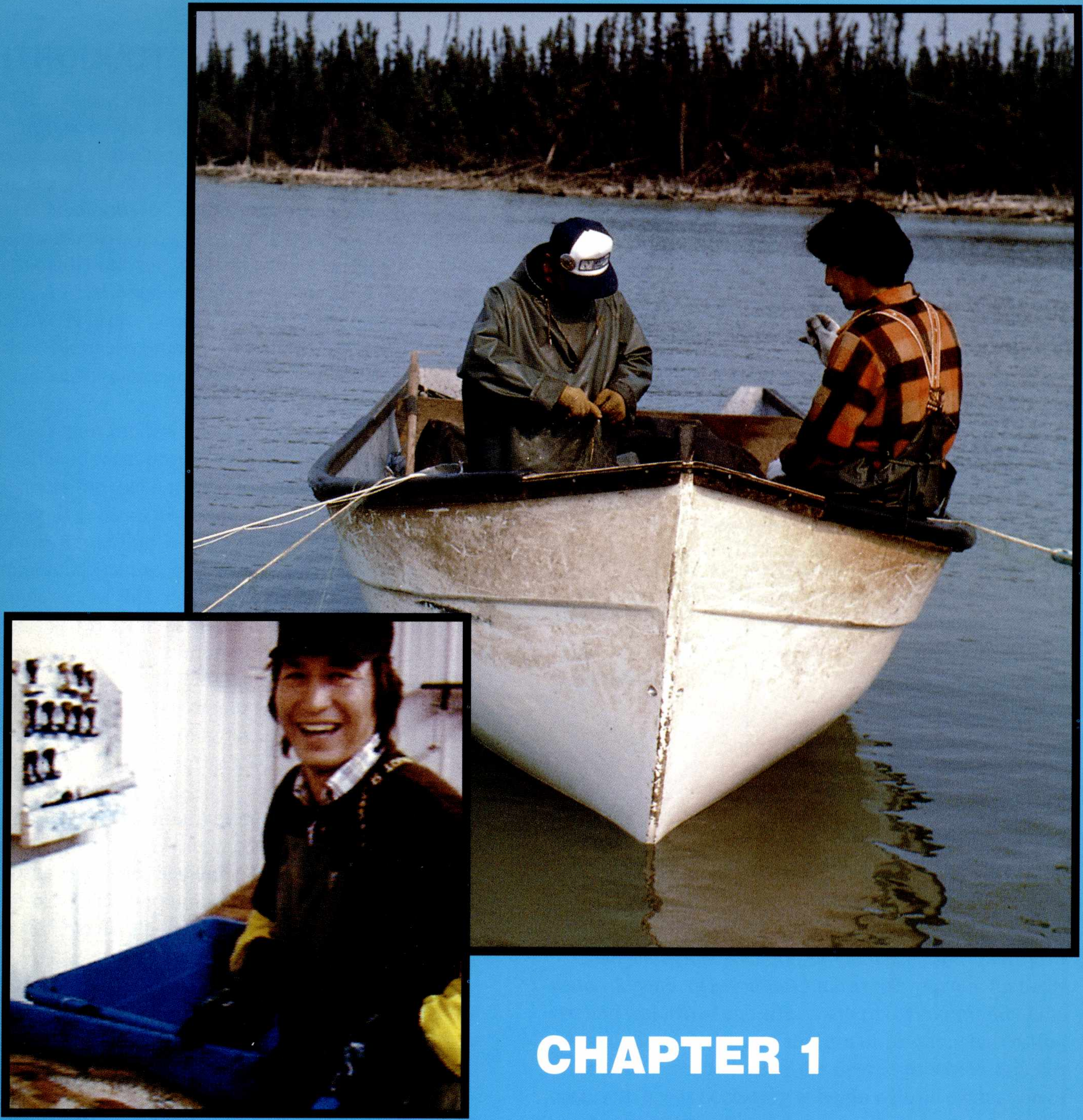


CHAPTER 1

RESOURCE HARVESTING

INTRODUCTION

FEMP focused on the physical, chemical and biological conditions in the FEMP study area; however, it was recognized that changes in these environmental conditions since the Lake Winnipeg, Churchill, Nelson (LWCN) Rivers Hydroelectric Project have also had major social and economic effects on the 6 native communities that traditionally utilized these waterways for livelihood and travel. A FEMP-sponsored study examined the effects of the LWCN project on the resource harvesting activities of the 5 Northern Flood Committee (NFC) communities (Usher and Weinstein 1991).

This report models resource harvesting systems, lists probable adverse project effects, and identifies impact indicators for social impact assessment of resource harvesting. It reviews available data for the five NFC communities regarding subsistence harvests, trapping, and commercial fishing, suggests the critical strengths and weaknesses of this database, and the requirements of a full social impact assessment. This report was intended to enhance our ability to monitor and predict the effects of the LWCN project and similar projects; it was not a social impact assessment per se, nor was it a substitute for one. Neither did it completely encapsulate all the resource harvesting activities of the NFC communities.

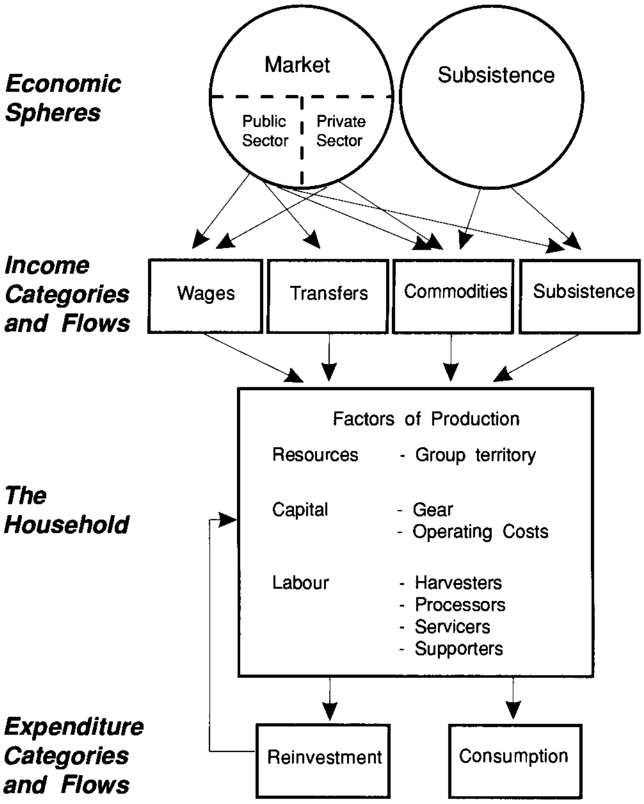
This chapter summarizes the results of the Usher and Weinstein report. It is augmented by information from a 1990 land use and occupancy study of South Indian Lake, a non-NFC native community located in the FEMP study area (Hrenchuk 1991).

SUBSISTENCE HARVESTING

Subsistence Activity

A distinctive characteristic of northern native communities is the place of resource harvesting as a central and integrative activity, and of subsistence as a socioeconomic system. Northern native community economies are recognized as being mixed, subsistence-based economies, in which the market and subsistence spheres are inextricably linked (Fig. 1.1). Subsistence, with its distinctive property and social relations, is an essential dynamic within the community culture and economy. Subsistence harvest and activity in this sense is not simply an addition to available income, but is inherent to the maintenance of social relations. Thus, subsistence-based economies may be affected by industrial development in a way which is fundamental, not peripheral, to their functioning.

Figure 1.1 The Household in the Mixed Economy, Market and Subsistence



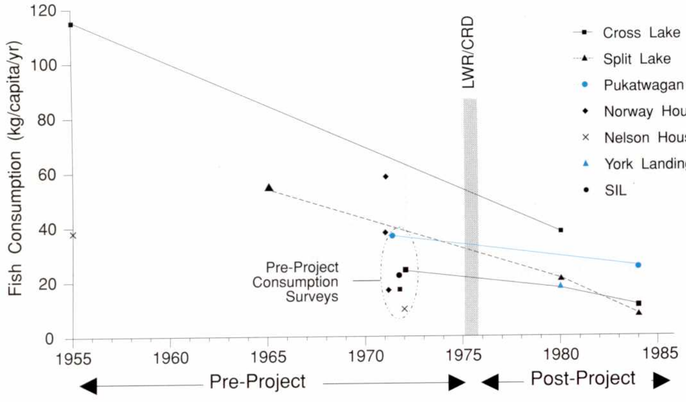
Changes in the location, abundance, and quality of water-dependent resources such as fish, aquatic fur bearers, and waterfowl can affect patterns of community land use and the viability and success of resource harvesting efforts. In order to assess the degree to which this has occurred, it was necessary to begin with an evaluation of subsistence harvesting data.

Subsistence Harvest Data

Data on household harvest by species, and if possible by edible weight, provide the most important indicator of subsistence income. Generally, these data result from surveys which are often done on the basis of informant recall. In the context of impact assessment, pre- and post-project data sets of adequate depth and consistency are required for comparative evaluation. It can be seen that data validity is dependent upon the extent to which standard methodological criteria have been followed and, furthermore, made explicit. By these standards, almost all of the existing estimates for northern Manitoba domestic harvests are of uncertain value.

For example, of the 21 estimates of per capita subsistence fish consumption for northern Manitoba cited by Usher and Weinstein, 11 were derived from surveys and 10 were speculative; 6 post-date the hydro project (Fig. 1.2). Many of these figures resulted from reiteration or extrapolation from results obtained through unacceptable and largely unspecified methods of earlier research. Thus any conclusions drawn from these data could not be regarded as robust. Furthermore, other potential indicators of subsistence fishing, such as participation, effort, and location are less easily quantified and are generally unavailable.

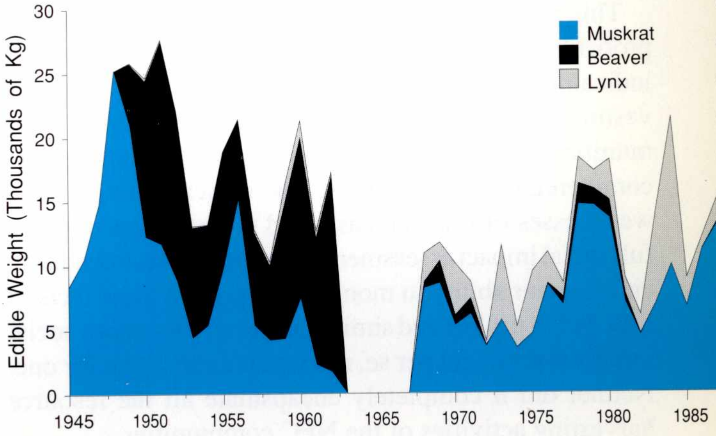
Figure 1.2 Annual Subsistence Fish Consumption (kg/capita) for Selected Northern Manitoba Communities, 1955 - 1984



Subsistence wildlife harvesting is comprised of big game, edible furbearers, waterfowl, and small game. Provincial records are the largest body of information about wildlife harvests, but they are of variable usefulness in evaluating subsistence harvesting. For example, conservation officer annual reports provide reasonably accurate data on pre-project ungulate harvest, and efforts have been made, in the 1980s, to acquire accurate data more systematically as part of the general ungulate management program in the province. However, the lack of accurate geographical references for the data on big game prevents the evaluation of the changes on a localized basis.

Edible furbearer harvests have been recorded indirectly through provincial records of furs sold, e.g. beaver, muskrat, and lynx. A recall harvest survey done in Split Lake and Cross Lake (Wagner 1985) provides a reference base for the provincial data. These statistics could be used to generate estimates of food harvest from edible furbearers over time (Fig. 1.3) and to evaluate pre- and post project harvests from affected and unaffected traplines.

Figure 1.3 Estimated Potential Food Harvests from Edible Furbearers Trapped on the Cross Lake Registered Trapline Section, 1945 - 1987



Pre- and post-project levels of subsistence harvesting of waterfowl and small game can not be compared because of lack of data.

Data on other potential indicators of subsistence wildlife harvesting activities (e.g. location, effort, and participation) are also lacking, for the 5 NFC communities. Some of this type of information is available for South Indian Lake from Hrenchuk's study. For example, an indication of the current level of participation in wildlife harvesting was given by the identification, from hunter interviews, of 232 prime sites for caribou, moose, muskrat and waterfowl in 39 of the community's 50 traplines.

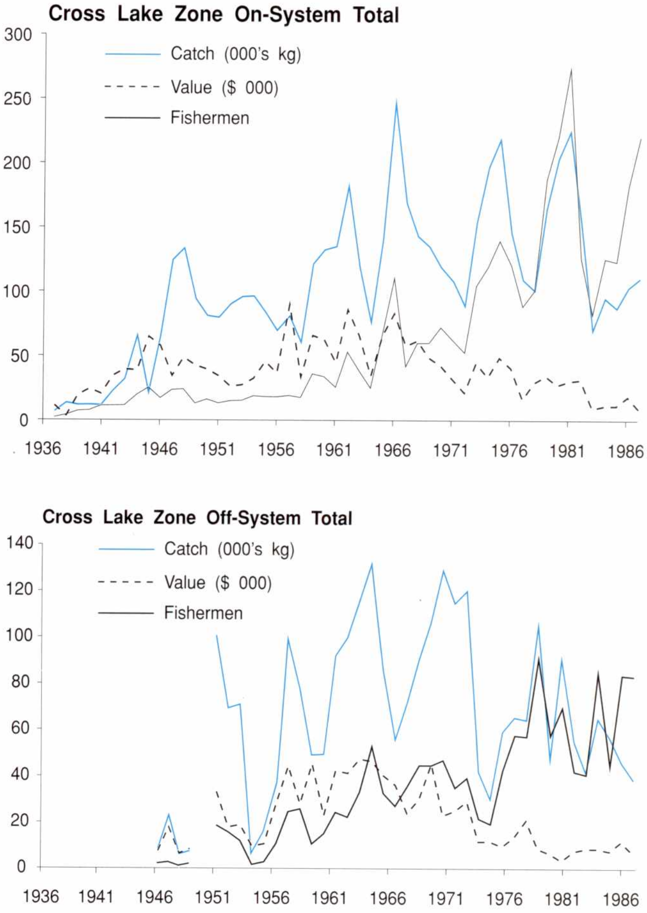
COMMERCIAL FISHING

When the modern commercial fishery was established in northern Manitoba, it was superimposed on the long-established Indian domestic fisheries. Since commercial fishing is strictly a licensed activity, in contrast to the case of subsistence harvesting, data have been collected on harvests since their inception. The first record of commercial fishing in the NFC area is from Sipiwesk Lake for the summer of 1937. The Southern Indian Lake fishery, the largest single fishery in northern Manitoba, began in 1942. Seventy-eight lakes were identified as having been commercially fished in the resource areas of the 5 NFC communities. Southern Indian Lake and 58 inland lakes have been fished commercially by South Indian Lake (Hrenchuk 1991).

Usher and Weinstein reviewed the available commercial data for its suitability in a social impact assessment, using: 1) weight of fish purchased; 2) dollar value to fishermen; and 3) number of licences issued, as indicators of volume, value, and participation respectively. On the basis of this review they concluded that the commercial fishing data were sufficiently comprehensive, uniform, and repetitive over many years to allow a preliminary assessment of the LWCN project's impact on catch and participation. These data indicated adverse impacts on the commercial fishing, for example: 1) the Cross Lake fishery experienced a sharp decline in production; 2) the Nelson House fishery has been partially contaminated by mercury; and 3) the cost of fishing increased at Norway House and possibly also at Split Lake. A more detailed economic valuation of the importance of commercial fishing in the FEMP study area could be assessed from the commercial fishing data, once appropriate adjustments to the data have been made to ensure constant dollars and comparable measures of weights. The adjusted values could be used as assessment indicators, by comparing data for on-system and off-system lakes (e.g. Fig. 1.4).

However, only some of the effects of the hydro project are revealed by the available commercial fishery data. For example, these data are inadequate for quantifying changes in cost and effort that can occur when fishermen respond to adverse biophysical effects by intensifying their efforts to maintain production, such as by setting more nets. Some of these changes in the Southern Indian Lake fishery have been documented by other FEMP sponsored studies that are reported on in chapter 2. Other changes, documented by Hrenchuk (1991), include low correspondence between pre- and post-impoundment preferred fishing sites, a reduction in the number of consistent net sites, and evidence of the diminution of lucrative pickerel fisheries.

Figure 1.4 Catch, Value, and Participation in the Cross Lake Commercial Fishery; for On-System and Off-System Lakes, 1936 - 1987



TRAPPING

Trapping has been integrated into the general pattern of resource harvesting of northern Manitoba communities since the time of European contact. Trapping allows for renewed contact with a land base, and for updating of the information base required for the successful harvest of wildlife. The opportunity to renew knowledge concerning the location, abundance, and movement of fur and non-fur species is essential in a hunting and trapping existence. Trapping is considered a traditional activity and, for many, it is part of a way of life. Although there are market and other economic considerations, the motivation for trapping in native communities does not primarily rely on profit. There is a desire to "come out ahead," but the provision of food or the fact of living in the bush may take priority.

Records of trapping harvests begin shortly after the inception of the registered trapline (RTL) program in the mid-1940s. There are about 226 traplines in the four RTL sections of the NFC communities, totalling 102,650 km^2. The South Indian Lake RTL section contains an additional 50 traplines with an area of approximately 35,000 km^2.

The fur record is split into three types: 1) 1950s, in which recorded fur sales are for the entire RTL; 2) 1960s to early 1980s, in which sales are recorded on the basis of individual traplines; and 3) 1984 onward, where a computerized record has been kept. Fur production and trapper participation data are available in considerable depth, both prior to and following the LWCN project (Fig. 1.5), and consequently may be useful in impact assessment. For example, the traplines can be aggregated by category of physical and biological impact, allowing the data to be used for hypothesis testing.

There are, however, potential difficulties in utilizing this record for impact assessment. Deviation from the record may occur where furs are not sold by the individual who trapped them, which seems likely within these community structures. Other uses, such as fur sales to tourists or furs used in clothing or craft manufacture, are excluded from the record. A major limitation in this data base, one that is shared with the commercial fishing data record, is that it contains no indicator of effort. Harvesters report that travel has become a major problem for some communities as a result of the LWCN project. However, increases in travel and cost are not measured in any institutional record-keeping system. The fur records are thus a necessary but not a sufficient tool for measuring change.

Additional data must be collected to supplement the fur data record in order to assess the importance of this activity and changes to it; one method would be trapper interviews such as those conducted in Hrenchuk's study of South Indian Lake. This study estimated that the sale of furs contributed approximately $206,000 in the 1987/88 season to the South Indian Lake economy, as well as providing furs for community use (e.g. crafts) and a source of meat, from the furbearers and from the animals hunted while in the process of trapping. The intensive and extensive use of the South Indian Lake resource area for trapping is evident from Fig. 1.6.

SOCIAL IMPACT ASSESSMENT AND RESOURCE HARVESTING

The social impact of industrial development on native communities, such as the LWCN project on the 6 native communities located in the FEMP study area, differs significantly from what its impact would be on southern Canadian communities, rural or urban. The content and pattern of social change itself, and the significance and distributional effects of this change, are distinctive in these communities because their socioeconomic characteristics, and the cultural and historical forces that created

Figure 1.5 Number of Registered Trappers and Value of Furs Harvested on Cross Lake, Norway House, Nelson House and Split Lake Registered Traplines, 1948 - 1988

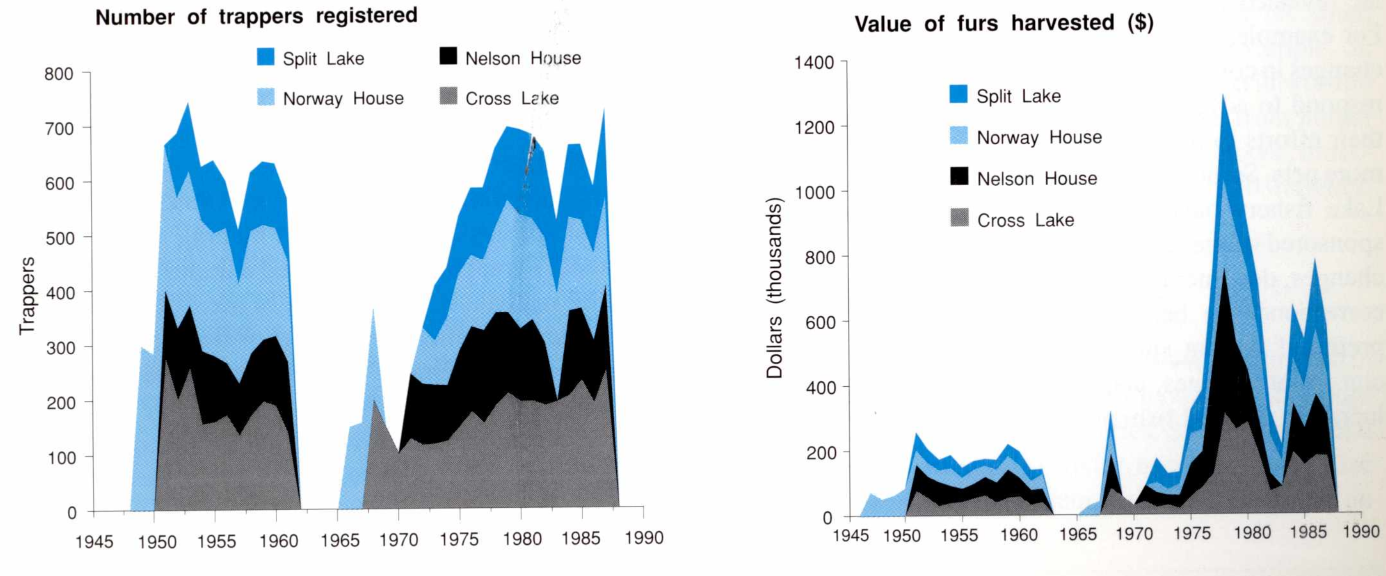
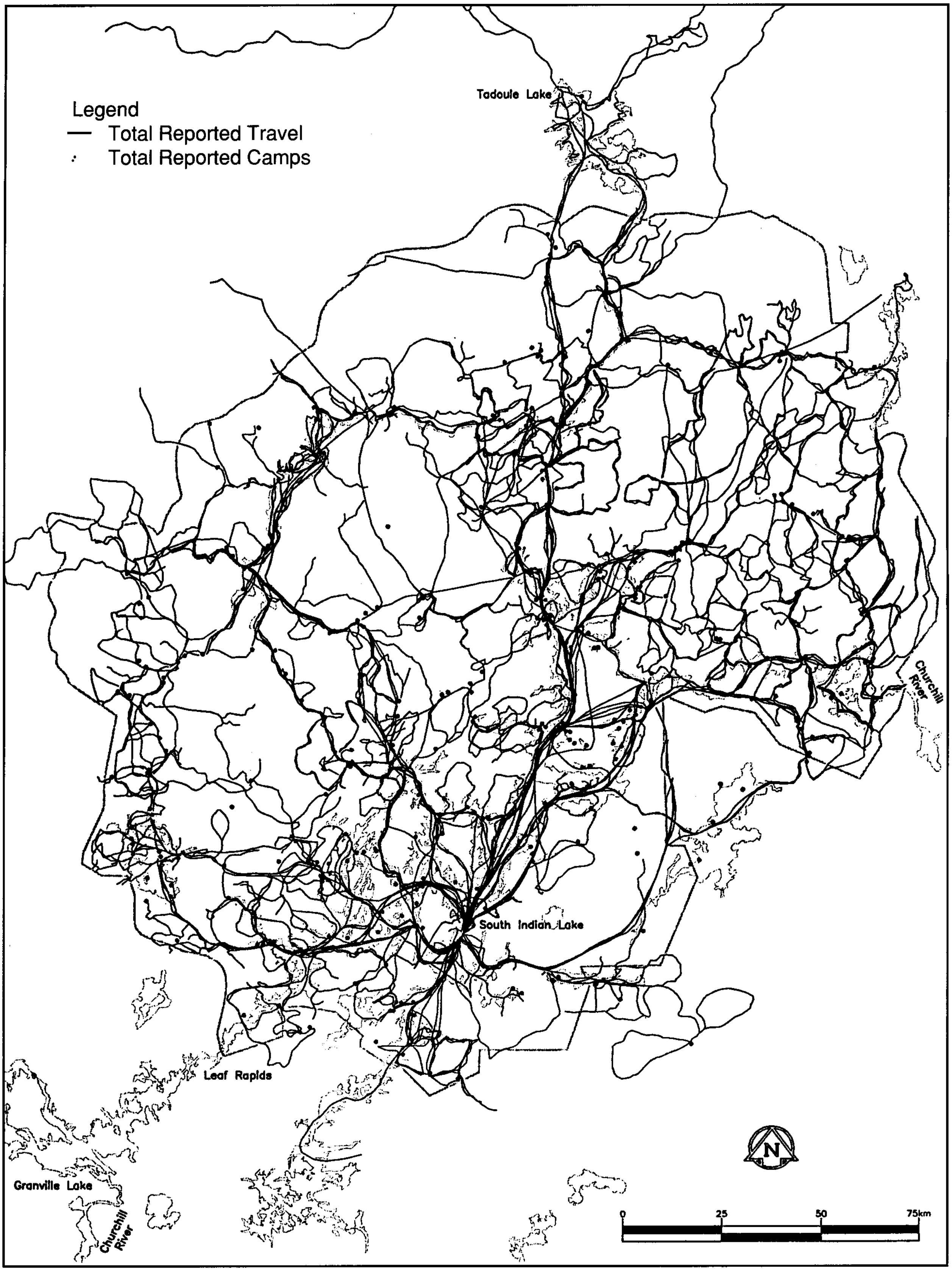


Figure 1.6 Total Reported Travel and Camps in the South Indian Lake Registered Trapline Area



them, are distinctive. Such differences, it is generally acknowledged, must be taken into account in social impact assessments (SIA).

However, this recognition in principle needs to be matched by more precise formulations of these differences in practice as the basis for selecting appropriate indicators. In the Usher and Weinstein study, institutional data on resource harvesting were reviewed to examine their potential as suitable social impact indicators. Interpretation of these data for this purpose, however, required three things: 1) a reliable model of resource harvesting systems (on their own and as part of a mixed subsistence-based economy); 2) sound historical knowledge of the evolution of these systems in each community (including their institutional manifestation in land tenure and resource management); and 3) an accurate record of the process and perception of changes induced by the LWCN project. These provide context as well as essential indications of significance, without which social impact cannot be assessed.

The two most readily available institutional data sets that can be used as impact indicators are those related to commercial fishing and trapping. These provide reasonably reliable indicators of harvest levels that can be identified (for the most part) at satisfactory levels of geographic detail, and which are of substantial historical depth. However, as noted, the fisheries data provide only limited information on harvesting effort, and the trapping data provide none at all. There are no other continuous records that can be used as impact indicators for resource harvesting. Estimates of subsistence harvesting are discontinuous in time and location, and idiosyncratic in method.

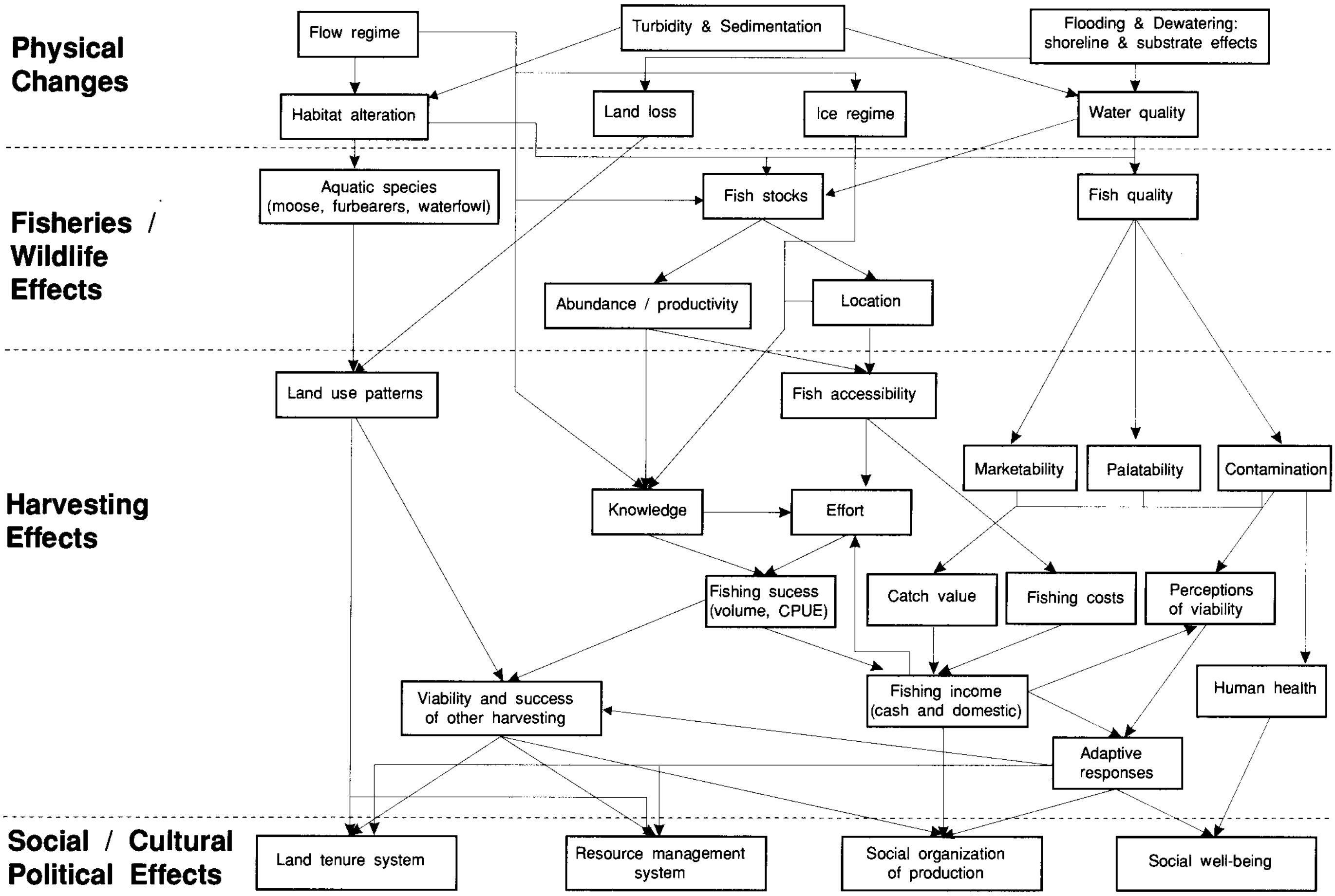
In view of the limited range of existing data, several additional lines of quantitative inquiry are required for social impact assessment. These include: 1) new or improved catch estimates for country food harvests, including fish, large and small game, and waterfowl; 2) improved effort estimates for the commercial fishery; 3) effort estimates for domestic fishing and hunting, and for trapping; 4) historic and recent patterns of land use and resource harvesting, geo-coded at least to the level of waterbody and trapline; and 5) dietary exposure to toxic substances (mercury in particular) in fish and other species, and changes in domestic consumption in response to perceived risks.

The successful employment of these formal, technical categories depends on them also being appropriate and recognizable categories in the minds of the affected harvesters. They cannot be imposed on each new situation on the sole grounds that they are technically sound or familiar. They must be discussed and pre-tested with harvesters. Harmonization of community and external technical categories requires disclosure and mutual understanding of their respective bases in experience, perception, and analysis.

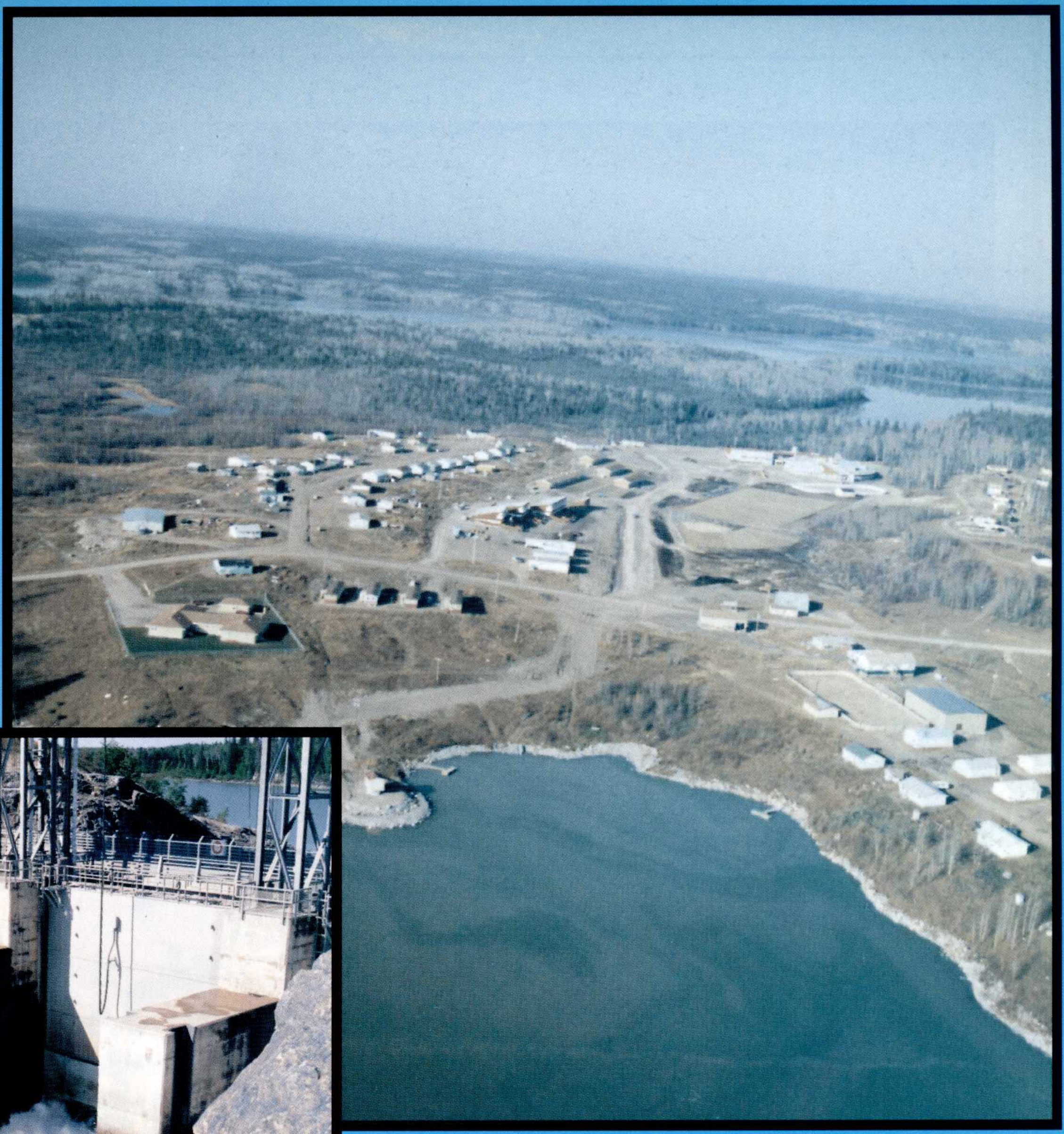
What is required is an account of harvesting now and in years past, of the changes in harvesting and of the explanations for these changes, and of the adaptive strategies by which people responded to them, to provide a context for interpreting the purely quantitative data. The type of information required includes: 1) the natural history of the land and the resources; 2) the land tenure and resource management system; 3) the social organization of production; 4) the functioning of the mixed, subsistence based-economy as a system; 5) the development of specific economic activities over time including their promotion and regulation by government; and 6) the emergence of reserve-based economics as enclaves. Finally, an assessment of the social impact of changes in resource harvesting and subsistence requires the compilation of standardized historical data sets of social indicators, coupled with documentation of community perceptions of impact.

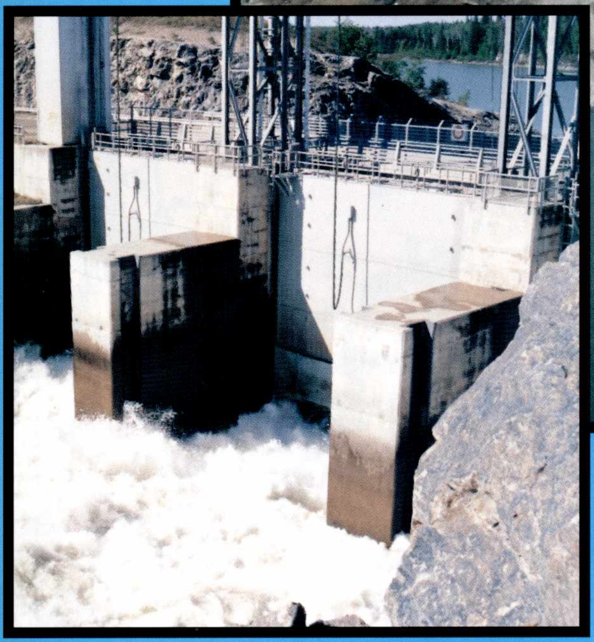
Meeting this entire range of data needs would require a combination of survey, extended interview, participant observation, and archival research. The scale and complexity of the hydro impacts (Fig. 1.7) and of the communities affected by them would require some innovative approaches, including those which focus on particularly important or indicative subsystems. SIA, as outlined here, also requires the use of control groups (e.g. neighbouring communities unaffected by the project), imperfect as they may be given the limited range from which to select. Also, the investigation of cause and effect relationships through such comparative analysis requires much more precise historical documentation than is found in conventional SIA. Finally, substantial effort would be required, at an early stage of any SIA research program, on the identification of significant variables and indicators and development of working definitions, with respect to both change phenomena and the hypothesized agents of change.

Figure 1.7 Probable Effects of the LWCN Project on Resource Harvesting and Subsistence Activities in the FEMP Study Area, with Emphasis on the Fishery



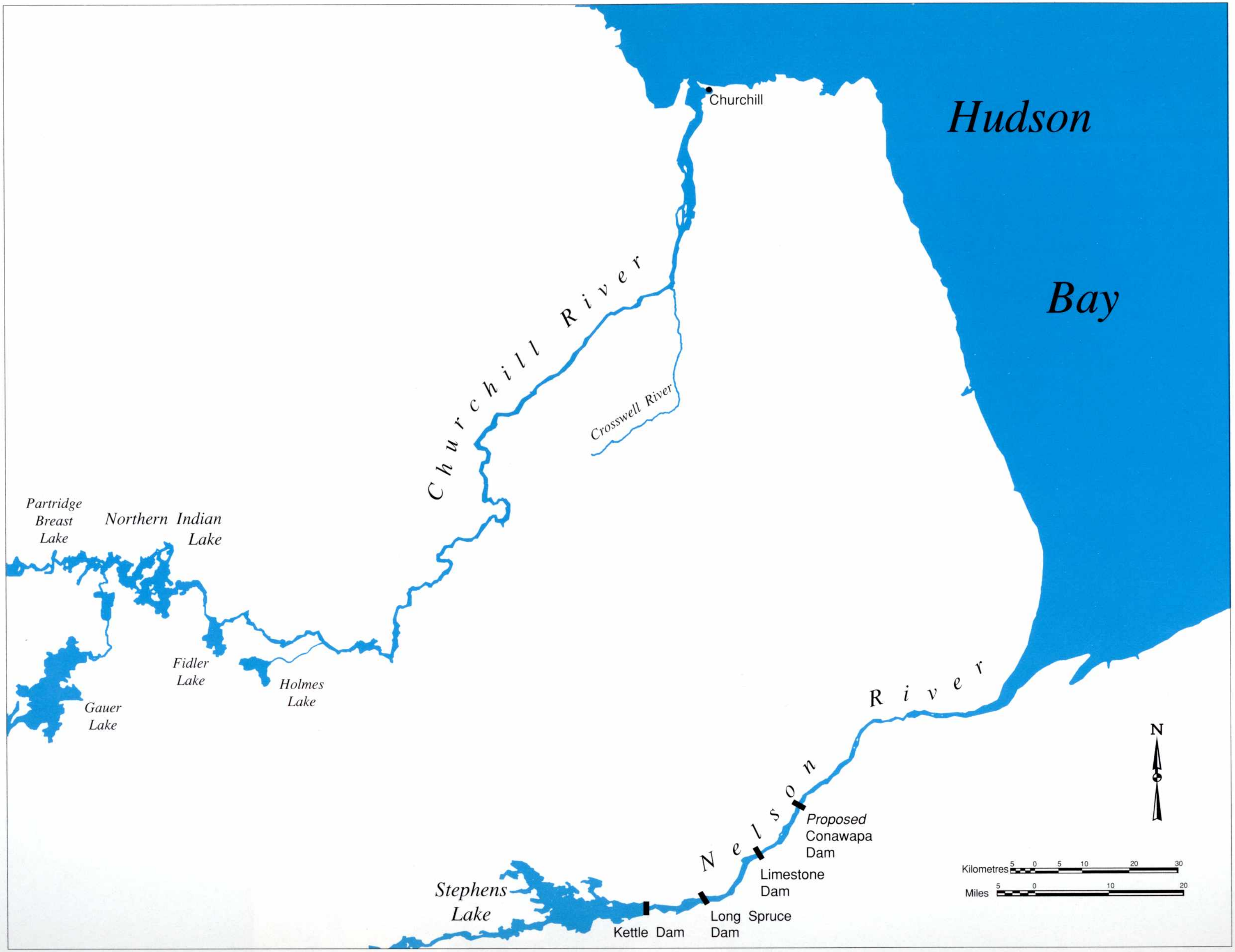
All of these considerations underline the importance, for a successful SIA, of research design and of extensive community involvement in that process. SIA must be community-based, not only in implementation and execution, but also in design and control. While external agencies can provide technical and financial resources, SIA which meets the requirements of public policy and scientific method will not occur until communities are institutionally enabled to initiate and direct it.





CHAPTER 2

SOUTHERN INDIAN LAKE AND THE CHURCHILL RIVER DIVERSION AREA


Hudson
Bay
Churchill
Churchill River
Crosswell River
Partridge Breast Lake
Northern Indian Lake
Fidler Lake
Holmes Lake
Gauer Lake
Nelson River
Proposed Conawapa Dam
Limestone Dam
Long Spruce Dam
Kettle Dam
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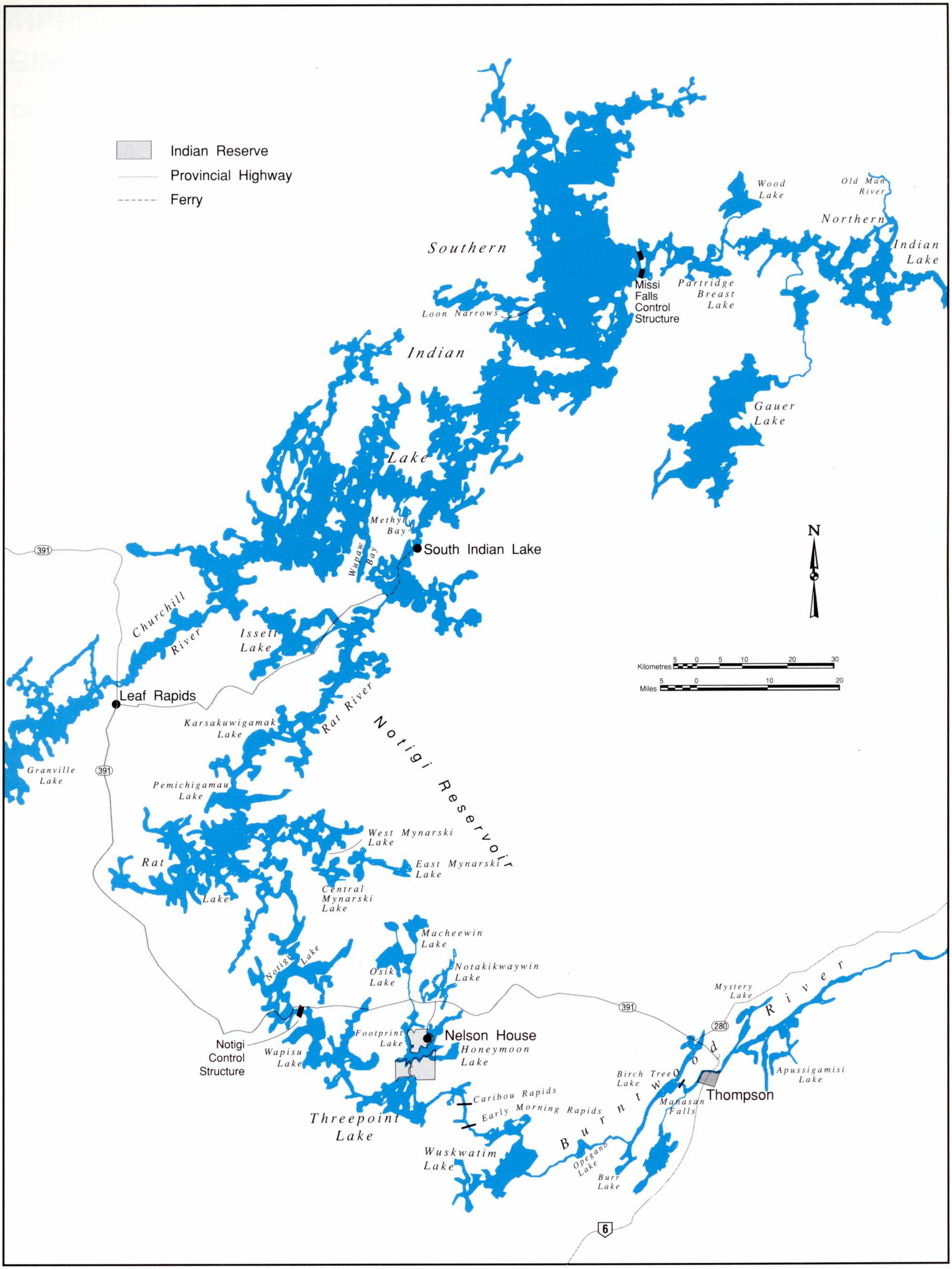
Indian Reserve
Provincial Highway
Ferry
Southern Indian Lake
Missi Falls Control Structure
Wood Lake
Old Man River
Northern Indian Lake
Partridge Breast Lake
Loon Narrows
Gauer Lake
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Wupaw Bay
South Indian Lake
N
Kilometres 5 0 5 10 20 30
Miles 5 0 10 20
391
Churchill River
Issett Lake
Leaf Rapids
Rat River
Karsakuwigamak Lake
Notigi Reservoir
Granville Lake
Pemichigamau Lake
West Mynarski Lake
East Mynarski Lake
Central Mynarski Lake
Rat Lake
Macheewin Lake
Notakikwaywin Lake
Notigi Lake
Osik Lake
Notigi Control Structure
Wapisu Lake
Footprint Lake
Nelson House
Honeymoon Lake
Mystery Lake
280
Burntwood River
Apussigamisi Lake
Birch Tree Lake
Manasan Falls
Thompson
Caribou Rapids
Early Morning Rapids
Threepoint Lake
Wuskwatim Lake
Opegano Lake
Burr Lake
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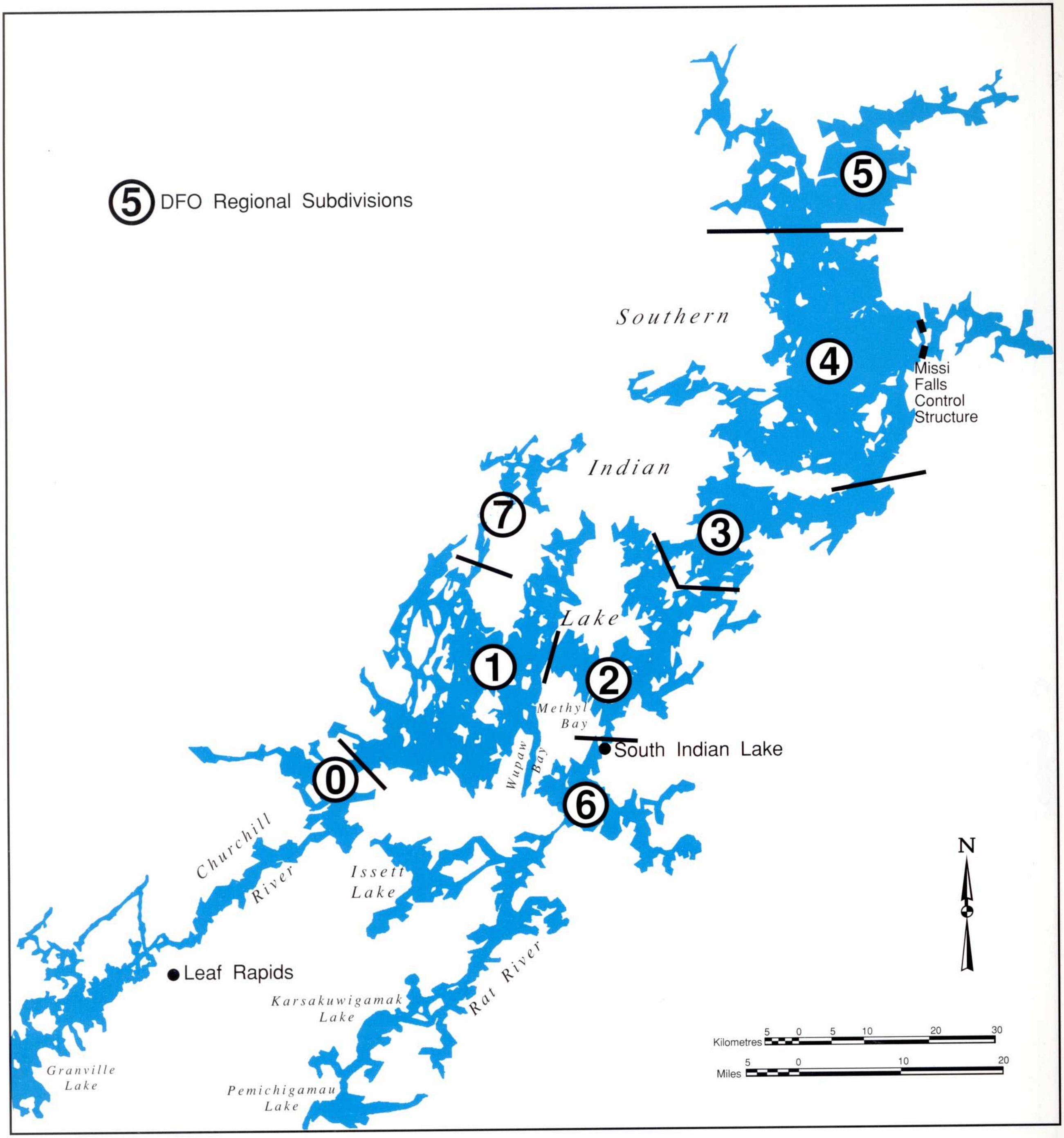

⑤ DFO Regional Subdivisions
Southern
Indian
Lake
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Missi Falls Control Structure
Methyl Bay
Wupaw Bay
South Indian Lake
Churchill River
Issett Lake
Leaf Rapids
Rat River
Karsakuwigamak Lake
Granville Lake
Pemichigamau Lake
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Kilometres 5 0 5 10 20 30
Miles 5 0 10 20

MORPHOLOGY AND SEDIMENT REGIMES

Introduction

The greatest changes in the morphology and sediment regimes in the FEMP study area occurred in the area affected by the Churchill River Diversion (CRD). Three complementary approaches were taken to improve our understanding of these changes, as follows: 1) a review of all relevant literature; 2) studies to determine the feasibility of assessing morphological changes; and 3) preparation of a sediment budget.

The literature review revealed that the information on the morphological conditions in the CRD area was limited. For example, a pre-project shoreline classification scheme (LWCN 1975) did not include a number of features that are critical to an impact assessment of the Lake Winnipeg, Churchill, Nelson (LWCN) Rivers Hydroelectric Project, such as inundated vegetation, floating organic material, cleared shorelines, or riprap. From the perspective of investigating sources of sediment production, this scheme is less than ideal due to its small scale, the lack of specific descriptions of shoreline stability (e.g. shoreline recession) or an assessment of the processes operating on the shoreline (e.g. wave action). More recent morphological investigations, which were based on the 1975 LWCN classification scheme, did not examine all of the CRD area nor did they quantify any of the observed morphological changes. Thus, it was decided to conduct two series of morphological feasibility studies, under FEMP, at a number of sites in the CRD area; one set categorized post-diversion shoreline characteristics and the other set focused on estimating sediment production at sites undergoing active erosion. Both sets of studies were based on air photo interpretation.

A number of investigations were pursued to enhance our knowledge of the sediment regime as follows: 1) collection of data on turbidity, non-filterable residue (NFR), and total dissolved solids (TDS) as part of the FEMP water quality program; 2) an assessment of sediment effects along the CRD; 3) preparation of an atlas of sediment sampling stations in the FEMP study area; and 4) an interagency comparison of field and laboratory procedures for determining suspended sediment concentrations. All of this sediment information, along with the results of the FEMP water budget (see volume 1, chapter 2), enabled the preparation of a sediment budget for the FEMP study area. A report on the sediment budget will be ready for publication in 1992.

This section summarizes the results of the FEMP investigations into the morphology and sediment regimes of the CRD area, by geographical area. A brief summary of the Department of Fisheries and Oceans' (DFO) pre-FEMP morphological and sediment investigations at Southern Indian Lake (SIL) is included for completeness. The section concludes with suggestions for future study.

The Lower Churchill River and Southern Indian Lake

Impoundment of Southern Indian Lake and diversion of the Churchill River down the Rat and Burntwood rivers greatly depleted the flows down the lower Churchill River. Comparison of 1969 and 1982/1984 air photos at 4 sites along the lower Churchill River showed an approximately 30% decrease in average channel width, resulting in: 1) desiccation of formerly extensive wetland areas; 2) exposure of large river bars and extensive areas of former river bed; 3) abandonment of former side channel areas; and 4) localized channel downcutting and bank erosion in former, apparently stable areas, causing locally significant sediment production.

The size of most tributary fans is now enlarged considerably. This appears to be caused by the exposure of previously submerged areas and, in a few locations, by the deposition of coarse textured materials transported by tributary streams. River depth has also been reduced but without comparative field surveys, the magnitude of this effect could not be determined.

In general, the observed morphological changes were relatively minor and progressing at very slow rates. This is primarily due to the many bedrock controls along the lower Churchill River and the incised nature of some channel reaches. Other factors contributing to the slow rates of morphological change are the many lakes along the mainstem river and its tributaries, which assure low rates of sediment transport and the harsh climate, which can result in slow rates of vegetation encroachment on to any abandoned active-channel zones.

While the changes in morphology in the lower Churchill River could have been predicted on the basis of experience from other case studies, in a qualitative manner, our ability to quantify the expected changes is far less certain, especially with respect to the prediction of subsequent physical and biological effects. Addressing these questions will require additional information such as: 1) the rate of sediment accumulation on tributary fans and the length of time for this process to stabilize; and 2) the percentage of side channels and wetland areas abandoned and changes in this percentage over time.

Immediately after the 3 meter impoundment of SIL in 1975/1976, the percentage of bedrock-controlled shorelines dropped from 76% to 14%. Most of the new shorelines were in fine grained, permanently frozen, ice-rich overburden consisting primarily of silt, clay and organics. As a result, there was considerable shoreline erosion; for example, erosion rates as high as 23 $m^3 \cdot m^{-1}$ of shoreline per year have been observed. (A predictive model for annual erosion rates at SIL was developed by DFO, based on the results from 20 shoreline monitoring sites, located in a variety of shoreline material and covering a wide range of wave exposure, height, and shoreline configurations. Because it automatically corrects for lake size, this model should be applicable to those lakes in the FEMP study area where the shoreline materials are similar to SIL, such as Footprint, Threepoint, and Wuskwatim. However, the SIL model has yet to be tested on other lakes.)

Most of the eroded mineral materials from SIL shorelines were deposited within a few hundred meters of the shore. However, there is evidence that these near shore deposits might be temporary and the sediments could eventually be remobilized and become part of the lake's general suspended load. At one site where shoreline erosion ceased after encountering bedrock, the recent near shore deposits disappeared over a period of 3 years.

The increased input of sediment into SIL raised sediment concentrations from about 5 $mg \cdot L^{-1}$, pre-impoundment values, to 30-50 $mg.L^{-1}$, post-impoundment. It also increased sediment output from SIL from about 120 000 tonnes in 1975, to 400 000 tonnes in 1976, 600 000 tonnes in 1977 and 550 000 tonnes in 1978. Most of this sediment outflow was transported to the upper reaches of the CRD, where much of it was lost in the upper Notigi Reservoir. Estimates of current sediment output from SIL will be available once the FEMP sediment budget is completed.

Notigi Reservoir Area

The Churchill River was diverted by means of a 9 km channel that was excavated from South Bay, at the southern end of Southern Indian Lake, to Issett Lake in the headwaters of the Rat River. In its natural state, the Rat River, like the Burntwood River, was mostly bedrock controlled, with stable shorelines. Lakes were an integral part of both rivers, with several river reaches consisting mainly of lakes, connected by short riverine segments. The Rat and Burntwood rivers are located in an area of low, undulating relief; vast, poorly drained areas of organic terrain are interspersed with some bedrock outcrops and other glacial features.

With the diversion of the Churchill River and the installation of the Notigi Control Structure, the series of small lakes along the Rat River was flooded to form Notigi Reservoir. The surface area of Notigi Reservoir, of 733 km^2, is substantially greater than the combined pre-development area of the incorporated lakes (e.g. Karsakuwigamak) of 182 km^2; the remaining area, 551 km^2, consists of flooded terrestrial terrain. Dead standing trees and floating peat bogs were prevalent through Notigi Reservoir immediately following diversion.

Notigi Reservoir, in particular the northern part of the reservoir embracing Karsakuwigamak and Pemichigamau lakes, is thought to be acting as a sediment sink, trapping most of the sediment output from SIL. In the lower part of the reservoir, formerly Notigi Lake, there is sediment deposition resulting from local shoreline erosion. There is some suggestion that shoreline erosion in this part of the reservoir may be increasing, as evidenced by the significantly greater turbidity levels in 1987 than those measured in the late 1970s. The decay and removal of inundated vegetation over time may be facilitating this apparent increase in erosion.

Footprint Lake Area

Interpretation of 1985 air photos for Footprint Lake and Honeymoon Lake, a small lake connected by flooding to Footprint Lake, showed that 37% of the combined shorelines contained standing debris, 47% had been cleared, and 55% were areas with floating vegetation or shallow water (Fig. 2.1). Analysis of 1972 air photos showed that approximately 75% of the shorelines at that time did not consist of any of the 5 categories used in the FEMP feasibility study (e.g. floating debris); these shorelines were categorized as unclassified. Consequently, it was not possible to conduct a comprehensive analysis of the changes from 1972 to 1985 in shoreline characteristics on the basis of this feasibility study. However, comparison of the classified shorelines showed the following changes: 1) an increase in shorelines with shallow water or floating vegetation from 14 km to 83 km; 2) an increase in shorelines with standing debris from 0.2 km to 56 km; 3) an increase in eroding shorelines from 4 km to 40 km, indicating the potential for widespread sediment production; and 4) an increase in cleared shorelines from 4 km to 71 km, the result of an attempt to reduce one negative impact of flooding.

Figure 2.1 Lengths of Coded Shoreline Types on Footprint, Honeymoon, and Threepoint Lakes, 1972 and 1985

	Footprint and Honeymoon Lakes				Threepoint Lake	
	1972		**1985**		**1985**	
Shoreline Type	**m**	**%**	**m**	**%**	**m**	**%**
C	410	0.5	8,800	5.8	0	0.0
CD	0	0.0	730	0.5	0	0.0
CDF	0	0.0	14,440	9.6	0	0.0
CE	3,930	5.2	7,120	4.7	1,540	0.8
CED	0	0.0	980	0.7	470	0.3
CEDF	0	0.0	330	0.2	0	0.0
CEF	0	0.0	3,880	2.6	1,700	0.9
CF	0	0.0	31,960	21.2	4,730	2.6
CR	0	0.0	90	0.1	0	0.0
CRE	0	0.0	2,600	1.7	0	0.0
D	0	0.0	7,040	4.7	27,510	15.1
DF	0	0.0	23,200	15.4	87,210	47.9
E	0	0.0	14,590	9.7	490	0.3
ED	230	0.3	7,580	5.0	39,870	21.9
EDF	0	0.0	1,430	0.9	3,380	1.9
EF	0	0.0	1,400	0.9	0	0.0
F	14,200	18.7	6,770	4.5	6,280	3.4
R	0	0.0	230	0.2	0	0.0
U	57,120	75.3	17,570	11.7	8,960	4.9

Classification System Used to Map the Distribution of Shore Zone Characteristics

C	Cleared of standing trees (debris may still be piled on the ground)
D	Standing inundated debris or vegetation in the nearshore or backshore area
F	Floating debris or organic matter in water; areas with shallow water depths
E	Eroding areas of shoreline as indicated by the absence of vegetation, the presence of foreshore sediment deposits, sediment plumes, mass-wasting features (such as rotational failures), etc.
R	Rip-rap
U	Unclassified shorelines not containing any of the above features.

NOTE:Shore zone characteristics are not mutually exclusive

A comparative analysis of the 1972 and 1985 air photos, along with topographic data, was used to generate quantitative estimates of sediment production at an actively eroding site, a small island near the southern shore of Footprint Lake, immediately east of Footprint River. It was found that the island's area decreased by approximately 6200 m^2 (from 9500 to 3300 m^2) and that the shoreline perimeter decreased by approximately 170 m (from 390 to 220 m). If it is assumed that 50% of the shoreline recession was due to erosion, rather than inundation, and that the depth of eroded sediment was 2 m, the total volume of material removed from this site would be in the order of 6000 m^3 or 27 m^3 per meter of present shoreline length. If it is further assumed that the majority of this erosion occurred in the 9-year period between 1977 and 1985, this would correspond to an annual rate of sediment production of 3 m^3 per meter of shoreline eroded. As both the shoreline heights and the percentage of bank recession due to erosion were likely under-estimated, the confirmed volume of material eroded from this site is a minimum estimate. Field investigations would be required to improve upon this estimate (e.g. determination of the ice content of the eroded shorelines).

It should be noted that the rate of sediment production at this island is not typical of all of Footprint Lake; for example other sites along the shoreline show no apparent change in shoreline position from 1972 to 1985. Photography taken in 1985 indicated generally much more turbid water conditions than in 1972, with localized sediment plumes extending off many of the west-facing shorelines. Comparison of 1972 and 1954 photography indicated no detectable change in shoreline position had occurred during this period; however there were some localized shoreline erosion sites, such as along the western shore of the first bay east of Footprint River.

Threepoint Lake

Analysis of 1985 air photos for Threepoint Lake showed that standing inundated vegetation or debris comprised almost 87% of the total lake shoreline; this is comparable to the total (84%) percentage of shorelines of Footprint and Honeymoon lakes that contained standing inundated vegetation or debris or that had been cleared. The percentage distribution of areas with floating vegetation or shallow water for Threepoint Lake was also similar to those of Footprint and Honeymoon lakes, with 56% and 55% respectively (Fig. 2.1).

Data on sediment related parameters such as turbidity, total dissolved solids (TDS) and non-filterable residue (NFR) were collected at 4 sites in the Threepoint Lake area as part of the FEMP water quality study. Fig. 2.2 summarizes these data, along with data on colour. (Colour is determined not by the presence of sediment, but rather by the presence of naturally occurring acids; these data are included in Fig. 2.2 because colour, like sediment, affects the aesthetic enjoyment of water.)

The post-development mean turbidity levels for the Burntwood River above Threepoint Lake were higher than for the other natural site in this area, the Footprint River above Footprint Lake; however, neither of the two natural sites showed any substantial difference from the pre-development period. Values for the natural Burntwood River site for the FEMP study period and the 1971 - 1974 pre-development period were 19 NTU and 14 NTU respectively, while the corresponding values for the natural Footprint River site were 9 NTU and 7 NTU.

The mean post-development turbidity levels for the Rat River above Threepoint Lake were marginally lower than for the natural Burntwood River site, while the mean post development turbidity value for the Burntwood River at the outlet of Threepoint Lake was intermediate between the two upstream sites. In contrast to the natural sites, there was an almost three-fold increase in mean turbidity levels at the Rat River site above Threepoint Lake, from 5 NTU in the 1971 - 1974 period to 17 NTU in the 1987 - 1989 FEMP study period. Lack of pre-development data for the Burntwood River site at the outlet of Threepoint Lake prevented a quantitative analysis of changes in turbidity here; however it is likely that turbidity levels also increased at this site.

Figure 2.2 Turbidity, Total Dissolved Solids, Non-Filterable Residue, and Colour for the Threepoint Lake Area, 1986 - 1989

Variable	Type	Units	Mean	SD	Max.	Min.
Burntwood River above Threepoint Lake [1]						
Turbidity	field	NTU	23.$^{+}$	7.	47.$^{+}$	12.$^{+}$
	lab	NTU	19.$^{+}$	4.	27.$^{+}$	13.$^{+}$
TDS		mg·L^{-1}	57.	10.	71.	25.
NFR	total	mg·L^{-1}	14.	7.	33.	6.
	fixed	mg·L^{-1}	11.	5.	27.	4.
Colour	true	relative	41.$^{+}$	10.	55.$^{+}$	25.$^{+}$
Rat River above Threepoint Lake [2]						
Turbidity	field	NTU	19.$^{+}$	8.	48.$^{+}$	9.$^{+}$
	lab	NTU	17.$^{+}$	8.	46.$^{+}$	11.$^{+}$
TDS		mg·L^{-1}	53.	4.	62.	46.
NFR	total	mg·L^{-1}	10.	7.	42.	5.
	fixed	mg·L^{-1}	9.	6.	38.	4.
Colour	true	relative	14.	4.	20.$^{+}$	5.
Footprint River above Footprint Lake [1]						
Turbidity	field	NTU	12.$^{+}$	6.	36.$^{+}$	6.$^{+}$
	lab	NTU	9.$^{+}$	3.	18.$^{+}$	4.
TDS		mg·L^{-1}	99.	26.	147.	41.
NFR	total	mg·L^{-1}	8.	4.	20.	4.
	fixed	mg·L^{-1}	6.	3.	16.	2.
Colour	true	relative	20.$^{+}$	8.	40.$^{+}$	10.
Burntwood River at outlet of Threepoint Lake [2]						
Turbidity	field	NTU	23.$^{+}$	25.	160.$^{+,++}$	10.$^{+}$
	lab	NTU	20.$^{+}$	21.	140.$^{+,++}$	8.$^{+}$
TDS		mg·L^{-1}	54.	4.	62.	47.
NFR	total	mg·L^{-1}	12.	21.	134.	3.
	fixed	mg·L^{-1}	11.	21.	126.	2.
Colour	true	relative	16.$^{+}$	5.	35.$^{+}$	10.

1 Sample period; May 1987 - Mar. 1989
2 Sample period; Sept. 1986 - Oct. 1989

Footnotes
$^{+}$ Exceeds limit for drinking water
$^{++}$ Exceeds limit for protection of aquatic life
Exceedences in the "Max." column indicates occasional exceedences.

Threepoint Lake to Wuskwatim Lake

The shoreline along the Burntwood River between Threepoint and Wuskwatim lakes is relatively steeply sloping while the valley flat is narrow and discontinuous; thus in most places the river is confined by the valley walls. An extensive series of bedrock controlled rapids is located in this section. Five segments of the Burntwood River in the vicinity of the major rapids (e.g. Caribou Rapids, Early Morning Rapids) were selected for air photo analysis; these segments focused on the most actively eroding sites along this reach of river.

Active channel areas were calculated for each channel segment from a series of air photos (Fig. 2.3). These results indicated that changes in surface areas between 1970 and 1972 were small and likely reflected the imprecision associated with the measurement technique as well as the slightly larger river discharge in 1972. Changes in area between 1970 and 1981 and 1985 were substantial, with active channel increasing in area by 23% and 36%, respectively. This increase in surface area reflected both the extensive shoreline erosion that occurred in the vicinity of the rapids and the inundation of low-lying sections of the valley flat and adjacent valley walls.

Figure 2.3 Estimated Changes in Active Channel Area and Volume of River Bank Erosion in the Vicinity of Gods, Caribou and Early Morning Rapids

Surface area of river in	*Total* (m^2)	*Change in surface area river in*	(m^2)
1970	633,000	1970-1972	9,500
1972	642,500	1970-1981	143,800
1981	776,800	1970-1985	230,200
1985	863,200	1981-1985	86,400

Length of actively eroding bank: 5,700 m

Total bank length: 10,960 m

Percent of bank actively eroding: 52%

Estimated increase in surface area due to bank erosion (m^2)	low	high
1970-1981	37,180	74,360
1981-1985	21,880	43,750

Estimated volume of eroded material (m^3)	low	high
1970-1981	74,360	297,440
1981-1985	43,760	175,000

Average annual sediment production (tonnes/year)	low	high
1977-1981	16,400	65,500
1981-1985	12,000	48,000

On the basis of air photo interpretation, it was estimated that approximately 50% of the total shoreline in the 5 river segments were actively eroding. Preliminary estimates of the volume of eroded material were calculated on the basis of certain assumptions (e.g. "low" and "high" estimates of the heights of eroding banks of 2 and 4 meters, respectively). On the basis of these assumptions, it was calculated that in the period 1977 to 1981, the average annual sediment load delivered to the diversion route from this reach was in the order of 1.6 to 6.5 x 10^4 tonnes per year. Calculated sediment delivery in the period 1981 to 1985 had a slightly smaller value of 1.2 to 4.8 x 10^4 tonnes per year.

Air photo analysis of Wuskwatim Lake indicated that no detectable change in shoreline position had occurred in the period between 1955 and 1972. Post-diversion photos taken in 1978 showed extensive sediment plumes along the near shore areas indicating that active erosion was occurring. Photography taken in 1985 indicated extensive areas of the southern end of Wuskwatim Lake were still undergoing active erosion. Eroded trees littered the near shore area and highly turbid water conditions occurred along the foreshore.

The volume of eroded material from low-lying sites was estimated to be approximately 6 m^3 per meter of shoreline. Shoreline retreat for shorelines with much higher bank heights was estimated to be slower than for shorelines from low-lying sites; however, given the larger amount of material which must be eroded from these higher relief sites, the inferred slower rates of bank retreat does not necessarily mean that smaller volumes of sediment are being produced from these areas. In comparison to southern Footprint Lake, shoreline erosion volumes may, nevertheless, be somewhat smaller, possibly reflecting both material characteristics and orientation of the lakeshore in relation to the prevailing wind direction.

As might be expected from the large estimated volumes of eroded material being produced along the diversion route, both at localized sites of active erosion such as rapids and the more widespread lakeshore erosion such as at Wuskwatim Lake, an examination of Landsat images from 1973 to 1986 suggest that the Burntwood River and on-route lakes downstream of Threepoint Lake are more turbid after diversion. In addition, there were some indications of increasing turbidity with distance along the diversion route. However, the large increase in river flows along the diversion route has meant that the post-diversion sediment concentrations are not greatly different from pre-diversion values; the major pre- to post-diversion change was in sediment tonnages, not concentration.

Thompson

No air photo interpretation was conducted of the Thompson area in the FEMP morphological feasibility study and consequently no preliminary assessment of the morphological changes is available for this site. However, this site has the best turbidity data record of any site in the FEMP study area, with data collected before diversion, during construction, and immediately after full diversion, as well as for the FEMP study period (Fig. 2.4 and 2.5).

Figure 2.4 Turbidity, Total Dissolved Solids, Non-Filterable Residue, and Colour for Thompson, 1987 - 1989

Variable	Type	Units	Mean	SD.	Max.	Min.
Turbidity	field	NTU	31.[+]	9.	65.[+,++]	18.[+]
	lab	NTU	27.[+]	6.	37.[+]	15.[+]
TDS		mg·L^{-1}	60.	4.	68.	54.
NFR	total	mg·L^{-1}	17.	5.	32.	7.
	fixed	mg·L^{-1}	15.	5.	29.	6.
Colour	true	relative	21.[+]	5.	30.[+]	10.

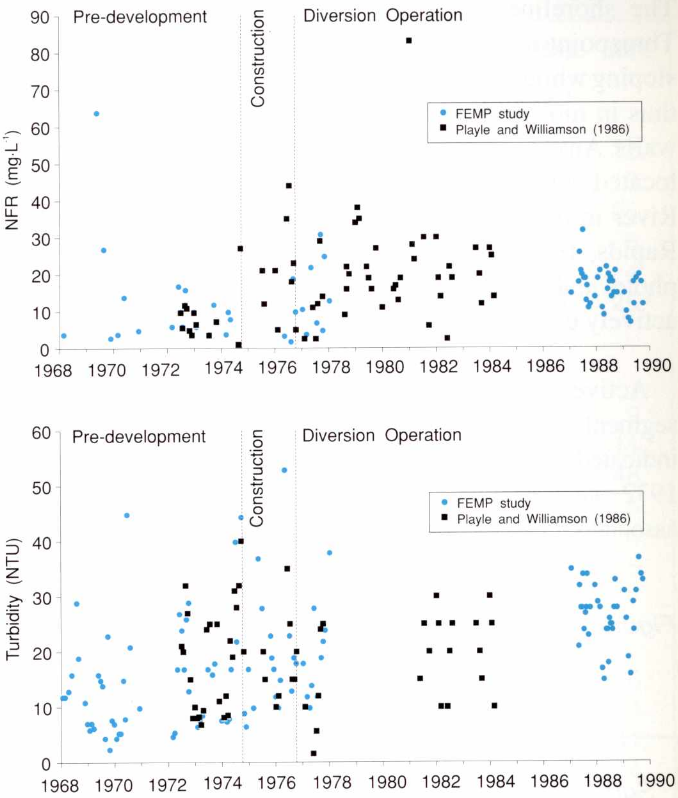
Footnotes

[+] Exceeds limit for drinking water

[++] Exceeds limit for protection of aquatic life
Exceedences in the "Max." column indicates occasional exceedences.

There was a 64% increase in turbidity levels during the construction period; possible contributing factors to this increase were the damming of the less turbid Rat River flows during this period and upstream construction activity. Immediately following the construction period (and the resumption of Rat River flows), there was a 21% reduction in turbidity, to levels slightly higher than the pre-diversion levels. Turbidity levels during the 1987 - 1989 FEMP study period were approximately 94% higher than the pre-diversion levels, due to increases in turbidity in the Rat River and the contribution from documented erosion sites along the Burntwood River downstream of Threepoint Lake.

Figure 2.5 Turbidity and Non-Filterable Residue for Thompson, 1968 - 1989



Data on non-filterable residue (NFR) are also available for the same period as the turbidity data. While these data show the same general trend over time as the turbidity data, statistical analysis of the NFR data did not show a significant increase, in contrast to the significant increase found for turbidity. One explanation for this apparent contradiction is the extreme variability in NFR compared to turbidity, particularly in the pre-development period (Fig. 2.5); the detection of a significant difference therefore requires a greater change in NFR than in turbidity. Turbidity may provide a better indication of the change in suspended sediment levels at this site because of the more extensive pre-development data record for this variable.

One interesting observation that is apparent from Fig. 2.5 is that both minimum NFR and minimum turbidity levels have increased since the diversion. Forty-two percent of the NFR measurements made before September 1976 were less than 7 mg·L^{-1}, the lowest NFR value between 1987 and 1989, while 58% of the turbidity measurements made in the pre-development period were less than 15 NTU, the lowest post-diversion turbidity level. This suggests that there has been an increased incidence of higher NFR and turbidity levels, at least at Thompson. It is not known how representative the Thompson site might be of the rest of the diversion route with respect to this apparent change since no other site has as extensive a data base; however, it may be a good indicator site because of its downstream location from some of the major erosion sites along the Burntwood River.

Future Conditions

In general, erosion to date has not been nearly as serious along the CRD as was predicted in pre-diversion studies. However, it might be overly optimistic to assume that erosion rates will necessarily diminish in the near future. As local erosional sites develop, an increasing length of bank tends to be exposed, so that volumetric rates of erosion could increase with time even if the rate of bank recession slows down. Furthermore, if gradual river bed degradation is proceeding at some points, banks that appear stable may eventually be undermined and begin to crumble. On the other hand, at some sites initial erosion slows or stops because bedrock becomes exposed as the overburden is removed.

A further severe complicating factor in projecting sediment concentrations and loads along the CRD is the fact that three further hydroelectric dams are planned for the Burntwood River, one at Wuskwatim and two further downstream. These three dams will alter water levels along most of the CRD between Notigi and First Rapids. In particular, most riverine reaches and all major rapids will be flooded and most lake levels will be increased significantly. Generally, fluvial erosion will tend to diminish as erosional sites are incorporated in reservoirs and head ponds; on the other hand, extended lengths of lake and reservoir shoreline susceptible to wind-wave erosion will be created. On balance, the net effect for most projected dams is estimated to be a further increase in sediment concentrations and loads.

Projection of future erosional rates and sediment production is subject to considerable uncertainty and could not be substantially improved without extensive field studies. Field studies would be necessary to enhance the accuracy of the analysis of the air photo interpretation, including those existing photos examined in the FEMP morphological feasibility study. In particular, field studies would be necessary to establish reference points for comparative analysis, to obtain more detailed information on shorelines (e.g. ice content) and to examine changes in channel morphology beneath the water's surface.

The participation of local residents in the establishment of the reference points as well as in the establishment of criteria for shoreline characterization would greatly enhance the accuracy and relevancy of the air photo interpretation, especially with respect to local concerns. An aerial video of the CRD, with a commentary by a local representative, would be invaluable in documenting the present and historical uses of the water bodies and their shorelines and how these uses have been affected by changes in shoreline characteristics.

The most cost efficient means of providing permanent documentation of post-diversion shoreline changes would be an ongoing program of aerial photography, flown at times of suitable lighting and river flow conditions and preferably at a 1:20,000 or larger scale. Recent developments in computer technology would enable cost-effective air photo analysis. For example, digital planimetric mapping techniques, in which shoreline locations and characteristics are stored in a computer based Geographic Information System (GIS) may allow multiple year comparisons to be undertaken relatively inexpensively. Similarly, recent developments in digital terrain mapping techniques and computer software will now allow computers to readily determine volumetric changes between two sets of topographic data, thereby computing the total volume of material eroded.

WATER QUALITY

Introduction

A three-year water quality study, from January 1987 to October 1989, of 61 physical and chemical water quality variables was conducted at 11 sites in the FEMP study area; 5 of these 11 sites were located in the Churchill River Diversion area. This section presents the results of the water quality study for the CRD area, as follows: 1) listings of those water quality variables whose values, from 1987 to 1989, exceeded the Canadian Water Quality Guidelines; 2) comparisons of current water quality conditions to historical conditions, to the extent that the data exist; 3) relationships, if any, between the water quality variables and river discharges; 4) differences in water quality conditions between the ice-free and ice-cover seasons; and 5) results of the limited bacteriological assessment of the recreational waters in and around Nelson House.

Canadian Water Quality Guidelines, established in 1987, were used as a means for assessing the water quality in the FEMP study area. Major uses of water for which Guidelines have been developed include drinking water, recreational use and the protection of aquatic life. Variations in environmental conditions across Canada will affect water quality in different ways; hence, many of the Guidelines may need to be modified according to these local conditions. The use of the Guidelines for assessing adverse effects in the FEMP study area is limited in that they have not been established for all variables, they are not based on site-specific cause and effect studies in northern Manitoba, and they do not take into account the effect of cumulative stresses caused when the values of two or more water quality variables simultaneously exceed the guideline values. They are thus a suggestive, rather than a definitive, basis for assessing adverse effects in the FEMP study area.

Natural Sites

Three of the 11 sample sites in the FEMP study area were located on unregulated rivers to provide a measure of current water quality in natural rivers in the study area and, where sufficient historical data existed, to determine the degree of natural variation in water quality between the pre- and post-development study periods. Two of the 3 natural sites were located in the Churchill River diversion area: Burntwood River above Threepoint Lake and Footprint River above Footprint Lake (Fig. 2.6).

Water quality in these two natural rivers was characterized by a number of variables which exceeded one or more of the Canadian Water Quality Guidelines for drinking water, recreational use, and protection of aquatic life. Mean levels of extractable aluminum (Al), total copper (Cu), extractable iron (Fe), turbidity and colour all exceeded the guidelines (Figs. 2.2 and 2.7). In addition, there were sporadic occasions when lead (Pb), cadmium (Cd), manganese (Mn), and zinc (Zn) exceeded the guidelines.

The natural rivers were distinguished by considerable stability in water quality between the 1972 - 1973 pre-development sampling period and the 1987 - 1989 FEMP study. There were no significant changes in the 17 variables for which comparable pre- and post-development data existed due to any cause other than a change in analytical methodology. Long-term temporal trends in metals could not be examined in the natural rivers; pre-development data on extractable or total metal species were essentially non-existent and FEMP data on dissolved metals were collected for too short a period (1 year) to enable statistical comparison.

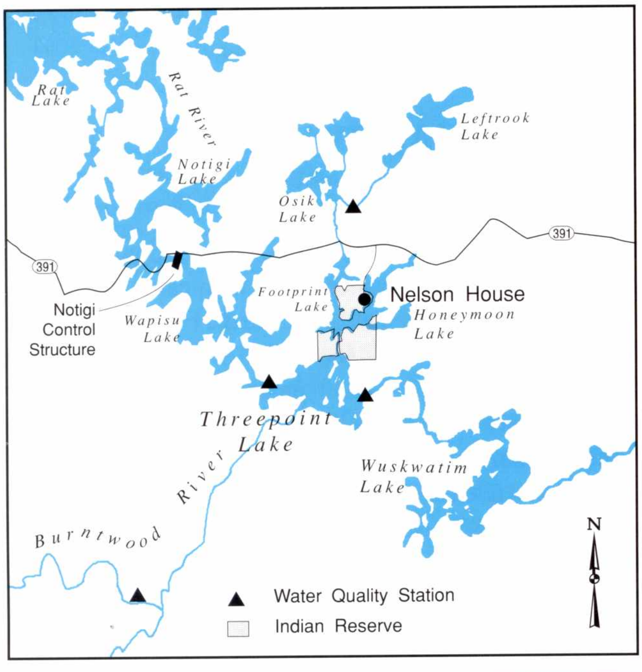


Figure 2.6 FEMP Water Quality Sites in the Churchill River Diversion Area

Figure 2.7 Water Quality Variables Which Exceeded the Canadian Guidelines along the Churchill River Diversion Route, 1987 - 1989

Var.	Type	Units	Mean	SD	Max.	Min.
Rat River above Threepoint Lake, Jan. 1987 to Sept. 1989						
Al	extractable	mg·L^{-1}	0.357^{++}	0.116	0.712^{++}	0.161^{++}
Cd	total	mg·L^{-1}	<0.0001	0.0001	0.0004^{++}	<0.0001
Cu	total	mg·L^{-1}	0.0053^{++}	0.0091	0.0400^{++}	0.0005
Fe	extractable	mg·L^{-1}	0.435$^{+,++}$	0.131	0.926$^{+,++}$	0.143
Pb	total	mg·L^{-1}	0.0008	0.0006	0.0021^{++}	<0.0007
Mn	extractable	mg·L^{-1}	0.019	0.013	0.081^{+}	0.004
Burntwood River above Threepoint Lake, May 1987 to March 1989						
Al	extractable	mg·L^{-1}	0.443^{++}	0.148	0.700^{++}	0.191^{++}
Cd	total	mg·L^{-1}	0.0001	0.0002	0.0009^{++}	<0.0001
Cu	total	mg·L^{-1}	0.0061^{++}	0.0081	0.0296^{++}	0.0012
Fe	extractable	mg·L^{-1}	0.642$^{+,++}$	0.165	1.050$^{+,++}$	0.183
Pb	total	mg·L^{-1}	0.0009	0.0007	0.0028^{++}	<0.0007
Mn	extractable	mg·L^{-1}	0.041	0.016	0.090^{+}	0.005
Footprint River above Footprint Lake, May 1987 to March 1989						
Al	extractable	mg·L^{-1}	0.190^{++}	0.097	0.410^{++}	<0.100
Cu	total	mg·L^{-1}	0.0232^{++}	0.1060	0.5530^{++}	0.0005
Fe	extractable	mg·L^{-1}	0.420$^{+,++}$	0.246	1.150$^{+,++}$	0.091
Pb	total	mg·L^{-1}	0.0011	0.0030	0.0161^{++}	<0.0007
Mn	extractable	mg·L^{-1}	0.057^{+}	0.078	0.296	<0.002
Zn	total	mg·L^{-1}	0.0040	0.0099	0.0515^{++}	0.0004
Burntwood River at the outlet to Threepoint Lake, Sept 1986 to Oct. 1989						
Al	extractable	mg·L^{-1}	0.377^{++}	0.209	1.360^{++}	<0.100
Cd	total	mg·L^{-1}	0.0001	0.0001	0.0006^{++}	<0.0001
Cu	total	mg·L^{-1}	0.0044^{++}	0.0059	0.0264^{++}	0.0005
Fe	extractable	mg·L^{-1}	0.477$^{+,++}$	0.303	2.010$^{+,++}$	0.135
Pb	total	mg·L^{-1}	0.0008	0.0009	0.0046^{++}	<0.0007
Mn	extractable	mg·L^{-1}	0.020	0.028	0.179^{+}	0.006
Burntwood River at Thompson, Jan. 1987 to Oct. 1989						
Al	extractable	mg·L^{-1}	0.491^{++}	0.154	0.826^{++}	0.230^{++}
Cu	total	mg·L^{-1}	0.0040^{++}	0.0051	0.0240^{++}	0.0009
Fe	extractable	mg·L^{-1}	0.624$^{+,++}$	0.189	0.931$^{+,++}$	0.256
Pb	total	mg·L^{-1}	0.0009	0.0006	0.0027^{++}	<0.0007

Footnotes

$^{+}$ Exceeds limit for drinking water

$^{++}$ Exceeds limit for protection of aquatic life

Exceedences in the "Max." column indicates occasional exceedences.

Water quality - discharge relationships for sites on the natural Burntwood and Footprint rivers were examined only for the FEMP study period, since there were no pre-development discharge data. General characteristics of the discharge relationships in the natural rivers included: 1) relatively large numbers of discharge correlated variables (the natural Burntwood River site had more variables correlated with discharge than at any of the other FEMP sites); 2) the dominance of negative slopes (e.g. conductivity, alkalinity, hardness, some major ions, etc. at the natural Footprint River site); and 3) the frequent occurrence of strong correlations (i.e. r^2 greater than 0.4).

At the natural Burntwood River site there were few differences in water quality between the ice-free and the ice-cover seasons. Concentrations of dissolved manganese (Mg), sodium (Na), and zinc (Zn) were higher under ice cover than during the open-water season, while colour and chlorophyll a were lower (the latter probably due to the imposition of light limitation on algae growth by accumulated snow cover). Before development there was more variation in these parameters, making any seasonal differences during this period less distinct.

At the natural Footprint River site, there were clear differences in the values for many variables between the ice-free and ice-cover seasons during the 1987-1989 FEMP sampling period (Fig. 2.8). Most variables were higher under ice cover (e.g. conductivity, alkalinity, hardness, most major ions, dissolved iron (Fe), etc.), while dissolved copper (Cu) and chlorophyll a were lower. A possible explanation for the higher levels of some variables under ice is their negative correlations with river discharge and the low flows which occurred on the river in winter. A qualitative examination of the limited pre-development database showed the same general pattern was present prior to development for those variables sampled in both the pre- and post-development periods.

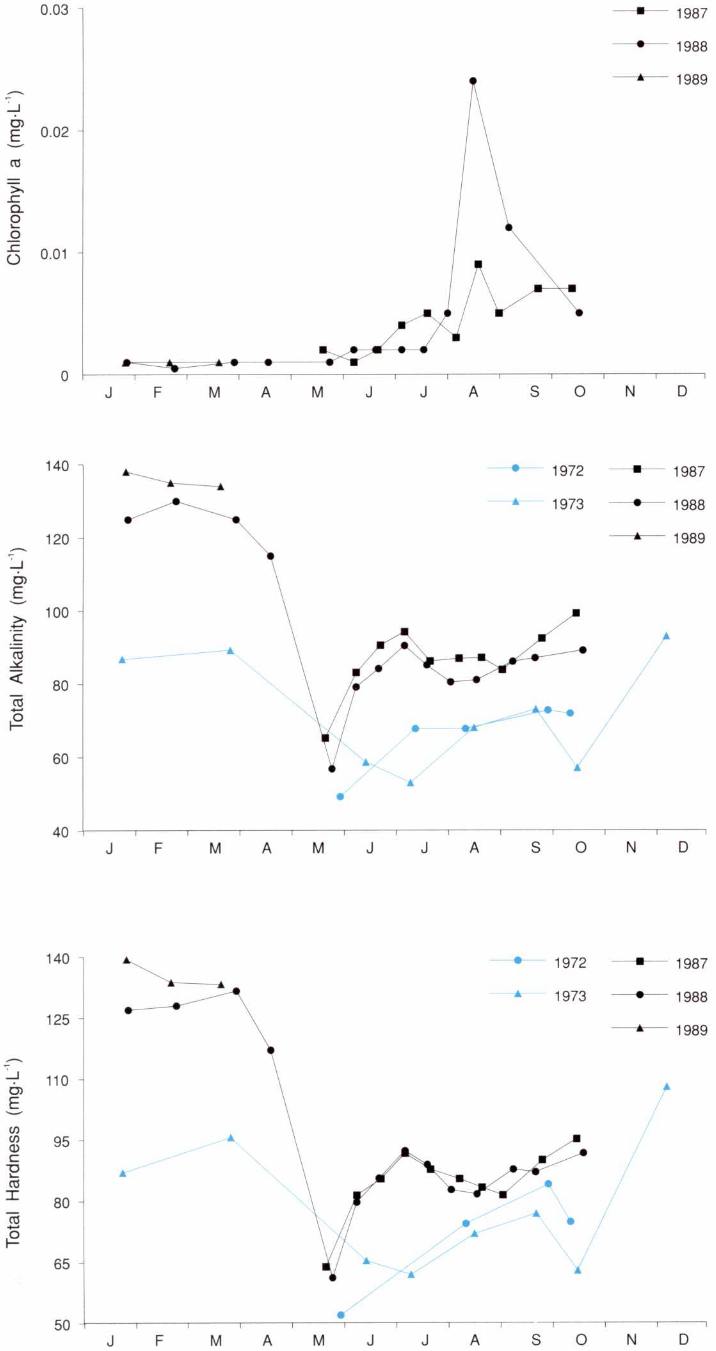


Figure 2.8 Seasonal Water Quality Trends for Selected Water Quality Variables in the Footprint River above Footprint Lake

Threepoint Lake and Nelson House Area

Two FEMP sample sites were located on regulated water bodies in this area: Rat River above Threepoint Lake and Burntwood River at the outlet of Threepoint Lake (Fig. 2.6). Water quality conditions at the Rat River site, during the FEMP 1987-1989 sampling period, were generally similar to those on the natural rivers: mean levels of extractable Al, extractable Fe, total Cu, and turbidity (and colour on the Burntwood River site) all exceeded Canadian Water Quality Guidelines, with sporadic occasions when Cd, Pb and Mn levels exceeded the guidelines (Figs. 2.2 and 2.7). Water quality conditions, during 1987 - 1989, for the Burntwood River at the outlet of Threepoint Lake were similar to those observed in the 3 rivers that flow into Threepoint Lake, namely: mean levels of extractable Al, extractable Fe, total Cu, and turbidity all exceeded Canadian Water Quality Guidelines, while there were sporadic occasions when Cd, Pb, and Mn levels exceeded the guidelines.

In addition to the 3-fold increase in turbidity at the Rat River above Threepoint Lake site, there were many other significant changes from the 1972 - 1973 pre-development sampling period to the FEMP study period (Fig. 2.9). Annual mean conductivity, alkalinity, hardness, total organic carbon (TOC), orthophosphate (PO_4), calcium (Ca) and reactive silicon (Si) decreased by 15 to 60%, while sodium (Na), potassium (K), chloride (Cl) and fluoride (F) increased by 10 - 80%. All of these changes are directly attributable to the diversion and are a reflection of differences in water quality between the Churchill River and the pre-existing Rat River. In most cases, the changes were no greater than the difference between the Rat and Churchill rivers.

There were two particularly interesting findings. The first was the greater decrease in TOC than what might have been expected; the flooded vegetation and organic soils in SIL and Notigi Reservoir might be expected to contribute organic carbon (C) to the water column. This issue will be examined more extensively by DFO. The second finding was the fact that total phosphorus (P) concentrations in the Rat River were not lower during the FEMP study than prior to diversion despite lower concentrations in the Churchill River. This finding suggests phosphorus continues to be

Figure 2.9 Comparison of Water Quality in the Rat River before (1972 - 1973) and after (1987 - 1989) Churchill River Diversion with Water Quality in the Churchill River near Leaf Rapids (1972 - 1973)

	Rat River		Churchill River
	Pre-Diversion	Post-Diversion	
pH (units)	7.9	7.66	7.6
Conductivity (µS·cm^{-1})	120	102	91
Turbidity (NTU)	5	17	5
Alkalinity (Total)	56.4	46.4	40.4
Hardness (Total)	61.2	45.9	41.9
TOC	10.7	6.38	8.7
NO_2NO_3	0.237	0.019	0.157
Ortho PO_4	0.005	0.002	0.003
Total P	0.023	0.026	0.017
Dissolved Na	2.2	2.7	3.0
Dissolved K	0.9	1.02	1.2
Dissolved Ca	18.2	12.3	12.5
Dissolved Cl	1.0	1.1	1.2
Dissolved F	0.005	0.09	0.08
Dissolved SO_4	2.7	2.5	3.6
Reactive Si	3.6	1.83	2.6
Dissolved Fe	0.07	0.096	0.05
Dissolved Mn	<0.01	0.003	<0.01
Dissolved Cu	0.002	0.0010	0.002
Dissolved Pb	<0.001	0.0007	<0.001
Dissolved Zn	0.007	0.0014	0.002

released from flooded terrain as late as 13 years after impoundment and diversion.

Impacts of diversion on water quality in the Burntwood River at the outlet of Threepoint Lake could not be quantified due to the lack of pre-development data for this site. Significant changes would be expected and should be qualitatively similar to those in the Rat River.

Positive correlations dominated the significant relationships between water quality and discharge, for the FEMP study period, for both the Rat River above Threepoint Lake and the Burntwood River at the outlet of Threepoint Lake. The dominance of positive correlations at these sites was the reverse of the situation in the natural rivers and in the Burntwood River at Thompson prior to diversion, where negative correlations dominated. This is evidence of the increased importance of eroded shoreline materials in regulating water quality. Whereas water quality typically was highest during periods of peak flow prior to diversion, water quality now decreases with increasing river flows. All of these correlations, however, were too weak to be of much use for impact prediction or management (i.e. r^2 less than 0.25).

Few variables displayed distinct differences between the ice-cover and ice-free seasons during the FEMP study. At the Rat River site, total dissolved nitrogen (TDN), total nitrogen (N) and dissolved NO_2-NO_3 occurred in higher concentrations under ice, while chlorophyll a concentrations were higher in the ice-free season (Fig. 2.10). This buildup of dissolved N constituents was likely due to lower dissolved N demand because of reduced algal growth rates under ice-cover. Higher TDN and dissolved NO_2-NO_3 concentrations under ice-cover were also observed at the Burntwood River site, probably due to the same cause as in the Rat River (Fig. 2.11). Much higher dissolved zinc (Zn) concentrations under ice-cover were also observed at this site. Higher dissolved Zn concentrations occurred under ice-cover in all three major inflows to Threepoint Lake as well, although the seasonality was not as distinct as at the outlet. Lack of data prevented a quantitative analysis of seasonal patterns in dissolved Zn in the pre-development period.

Bacteriological Assessment

A bacteriological assessment of the recreational water in the Nelson House area was conducted in 1987, consisting of a seasonal study from May to September at 8 sites and an intensive study in the vicinity of the sewage treatment plant outfall in August. Geometric mean fecal coliform bacteria counts for the seasonal study generally ranged between 1 and 6 bacteria per 100 millilitres (mL). The geometric mean for the intensive study was 38 per 100 mL; a few high coliform counts were recorded, but they were transient. Comparison of the results of the intensive study to the Canadian Water Quality Guidelines of 200 bacteria per 100 mL indicates that the bacteriological water quality at this site was generally acceptable for recreational water uses. Limited sampling frequency for the seasonal study precludes direct comparison of its results to this guideline.

Burntwood River at Thompson

Water quality in the Burntwood River near Thompson had similar characteristics to the other sites in the local (Burntwood, Footprint, and Aiken rivers) and Churchill River drainages. Levels of most parameters were well

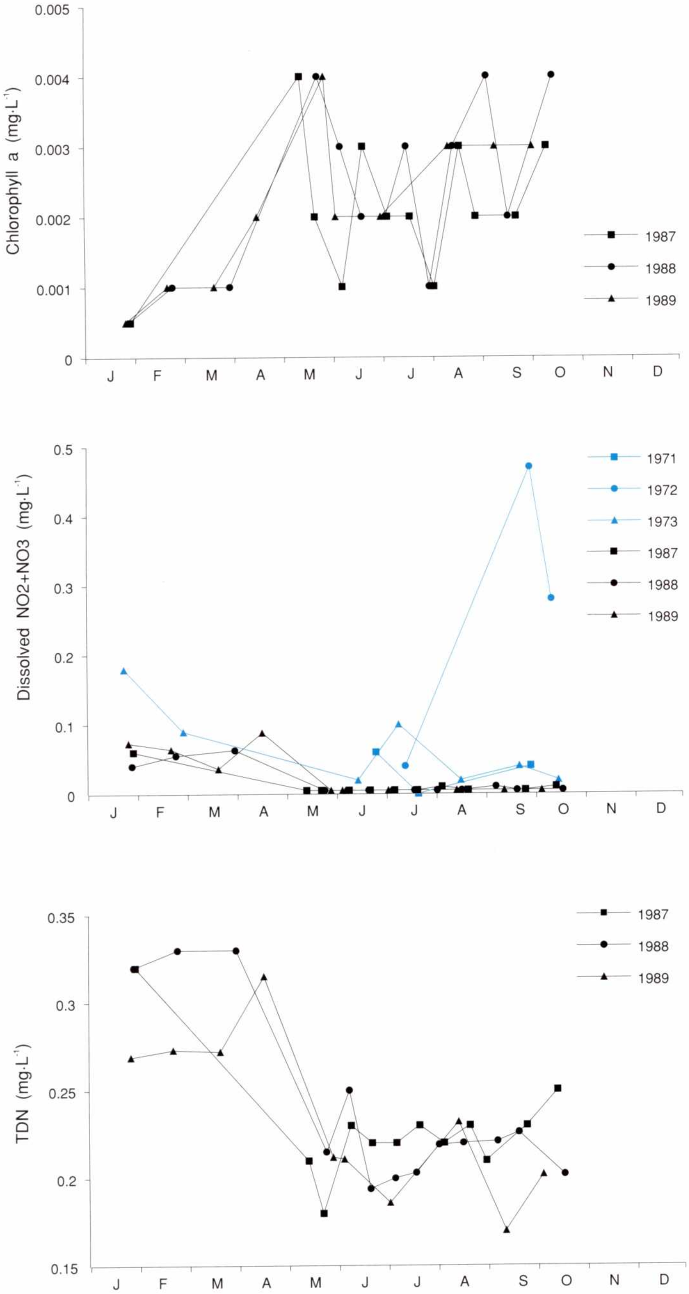


Figure 2.10 *Seasonal Water Quality Trends for Selected Water Quality Variables in the Rat River above Threepoint Lake*

within Canadian Water Quality Guidelines, and those which exceeded the Guidelines were the same as in the local drainages. Mean values for turbidity, colour, extractable Al, total Cu, and extractable Fe all exceeded Canadian Water Quality Guidelines, and there were sporadic occasions when Pb exceeded the guidelines (Figs. 2.4 and 2.7).

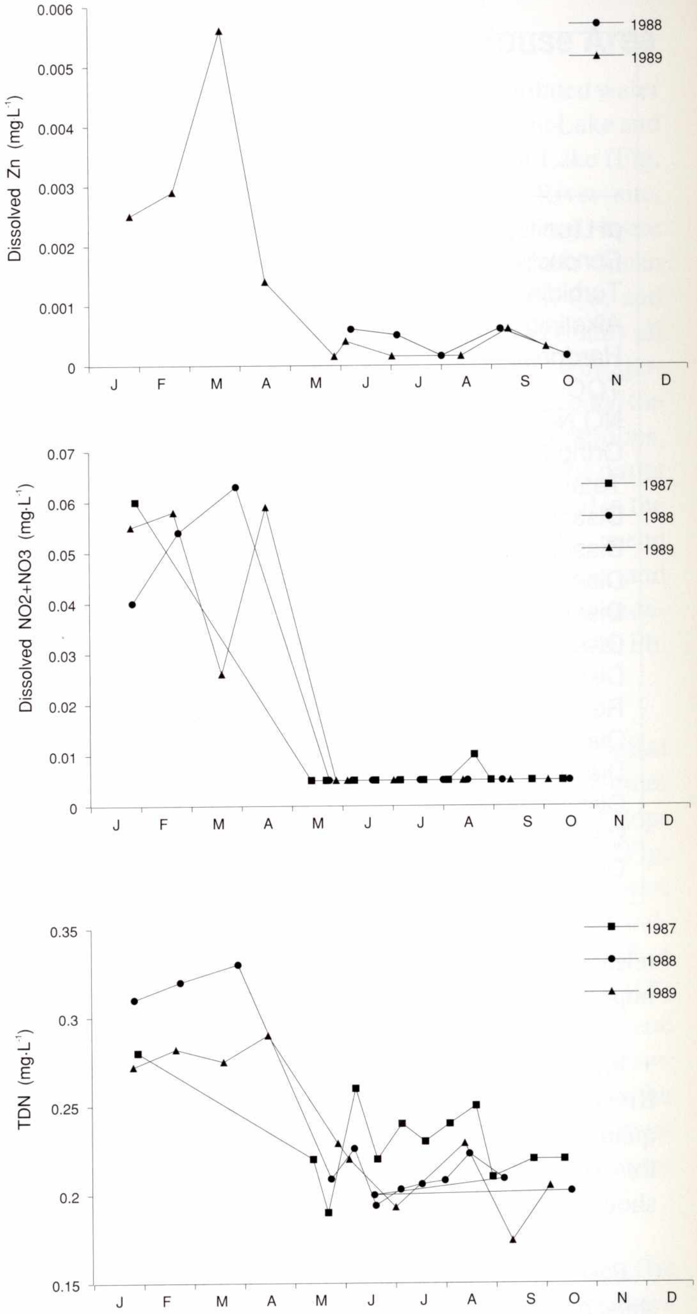


Figure 2.11 *Seasonal Water Quality Trends for Selected Water Quality Variables in the Burntwood River at the Outlet of Threepoint Lake*

The Burntwood River at Thompson is the only site in the FEMP study area which was sampled before, during, and immediately after construction of a hydro project, as well as during FEMP; it also has the most extensive baseline data record, with sampling conducted continuously from 1967 to 1974. However, only 10 variables were common to all 4 periods, thus seriously limiting the nature of the water quality changes which can be conclusively stated.

Most of the changes in water quality did not occur until after the autumn of 1977, when the diversion was fully operational. Significant water quality changes that occurred between the pre-development (1967 - 1974) period and the FEMP (1987 - 1989) period included decreases in conductivity, alkalinity, hardness, reactive Si, dissolved Ca, and dissolved Fe, and an approximate doubling of turbidity (Fig. 2.12). Several variables were not routinely measured until the construction period, 1974 - 1976; among these significant reductions occurred in dissolved Mg and in extractable Al between 1974 - 1976 and 1987 - 1989. All of these decreases were due to the introduction of softer Churchill River water (Fig. 2.9).

There were several changes in water quality that were evident during the construction period, when Rat River flows were cut off at the outlet of Notigi Lake or during the immediate post-diversion period (1976 - 1977), as flows were gradually increased. For example, hardness increased during the construction period, due to reduced flows on the Burntwood River, returned to the pre-development level in the immediate post-diversion period, and further declined after 1978. Another example was the approximate doubling of total phosphorus (P) concentrations between the construction and immediate post-impoundment periods, probably due to the release of P from flooded organic soils and vegetation. The total P concentrations in 1987 - 1989 were not significantly different from the levels prior to diversion (1967 - 1974); however, the P flux may still be occurring, as indicated by higher total P concentrations in the Rat and Burntwood rivers than in the Churchill River. The end point of this process is not the pre-development concentration for the site, but the level in the source of the flow, the Churchill River.

Most correlations between water quality and discharge during the pre-development period and all significant correlations during the construction and immediate post-construction periods were negative; however, all significant correlations with discharge during the FEMP study were positive at this site. Although more variables were correlated with discharge during the FEMP study, the relationships generally were much weaker than in the preceding periods and thus do not offer much promise for impact prediction or management. It is interesting to note that turbidity was positively correlated with discharge in the pre-development period, was not correlated with discharge in the construction and immediate post-diversion periods, and then was again positively correlated with discharge in the FEMP study period. The occurrence of positive correlations with discharge for turbidity and NFR in the Burntwood River at Thompson but not upstream at the outlet of Threepoint Lake is indicative of shoreline erosion occurring between these points. The discharge relationships for turbidity and NFR are strong enough to offer some promise of predictive value.

Figure 2.12 Means and Standard Deviations for Ten Water Quality Variables at Thompson, 1967 - 1989

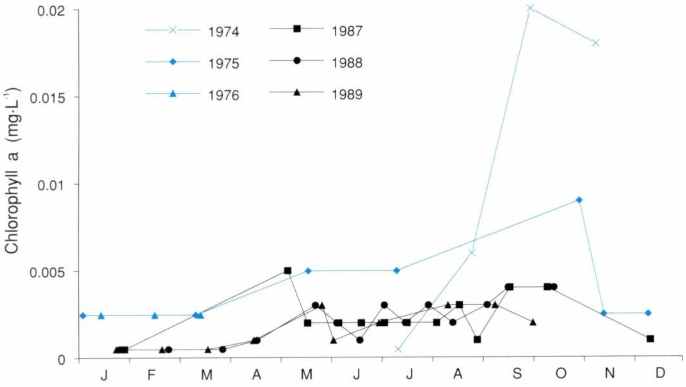
	Conductivity (lab)		**Turbidity (lab)**		**Alkalinity (total)**		**Diss Fe**		**Reactive Si**	
	mean	s.d.	mean	s.d.	mean	s.d.	mean	s.d.	mean	s.d.
Pre	141.0[1]	7	14.0[1]	2	65.0[1]	3	0.138	0.027	3.94[1]	0.25
Const	148.0	6	23.0	1	70.0[1]	4	0.150	id.	4.10	0.76
Post	125.0	25	18.0	<1	60.0[1]	2	id.		3.25[1]	0.50
Current	115.0	2	27.0[1]	2	53.0[1]	1	0.077	0.010	2.01	0.12

	Diss Ca		**Hardness (total)**		**Diss Mg**		**Extr Al**		**Total P**	
	mean	s.d.	mean	s.d.	mean	s.d.	mean	s.d.	mean	s.d.
Pre	21.3	2.7	69.0[1]	3	id.		id.		0.028	0.009
Const	21.5	0.8	77.0[1]	4	6.5[1]	0.1	0.870	0.207	0.039	0.010
Post	id.		66.0[1]	<1	id.		0.545	0.021	0.065	0.007
Current	14.6	0.1	53.0	<1	4.0	0.1	0.488	0.095	0.036[1]	0.003

Pre = 1967 to May 1974
Const = June 1974 to August 1976
Post = September 1976 to December 1977
Current = January 1987 to October 1989
id. = insufficient data
[1] = discharge - standardized variables

As was the case at many other sites, chlorophyll a concentrations were generally lower under ice cover than during the ice-free season (Fig. 2.13); this probably reflects the reduced light penetration through the snow and ice. The extensive data base for the Burntwood River at Thompson site, including the routine measurements under ice cover in the pre-development and construction periods, enabled the most complete analysis for seasonal trends of any of the FEMP water quality sites. Only chlorophyll a showed a marked seasonality in either of these two periods.

Figure 2.13 Seasonal Trends in Chlorophyll a Concentrations in the Burntwood River at Thompson



MERCURY

Introduction

Mercury in the FEMP study area first became an issue in the late 1970s with the discovery of elevated mercury levels in fish along the CRD route. A federal-provincial study, The Canada-Manitoba Agreement on the Study and Monitoring of Mercury in the Churchill River Diversion (CMMA), concluded that extensive flooding of organic material led to the increased net production of methyl mercury by bacterial processes and a subsequent increase in methyl mercury levels in fish. It was estimated that it would take decades before there would be a return to pre-development mercury levels. Recommendations were made for an ongoing fish monitoring program, for the implementation of fish management activities as a means of mitigating the mercury problem, and for the explicit consideration of elevated mercury levels in fish following impoundment in the planning and costing of reservoirs.

The mercury problem was not a predicted impact of the CRD, yet it is among the most severe of the impacts which have been documented and a continuing source of concern for the native residents within the FEMP study area. Because of the importance and expected longevity of this problem, continued investigation of mercury became a FEMP priority. In particular, the goals of the FEMP mercury program were: 1) to better define the expected duration of the mercury problem; 2) to provide a better understanding of the factors affecting the net production of methyl mercury to determine if it could be regulated; and 3) to monitor mercury levels in fish in Southern Indian Lake (SIL) and Issett Lake to identify any recent temporal trends. (Mercury levels in fish in other major lakes in the FEMP study area were monitored by the Manitoba Department of Natural Resources.)

To better define the expected duration of the mercury problem, measurements of net mercury methylation rates were made at a number of reservoirs of different ages: 1) Laurie Reservoir (flooded in the early 1950s); 2) Notigi Reservoir (flooded in 1976); 3) Sipiwesk Lake (flooded in the early 1960s); 4) Stephens Lake (flooded in the early 1970s); and 5) Sokatisewin Lake, Saskatchewan (Island Falls Reservoir, flooded in 1929). In addition, yearly measurements of net mercury methylation rates were made, from 1986 to 1989, at two locations; Methyl Bay, a small isolated bay of SIL and Granville Lake, a riverine lake on the Churchill River immediately upstream of SIL which has not been affected by the flooding. For all locations, 5 surveys were taken, at approximately 2 week intervals, from 3 or 4 near shore and offshore sites.

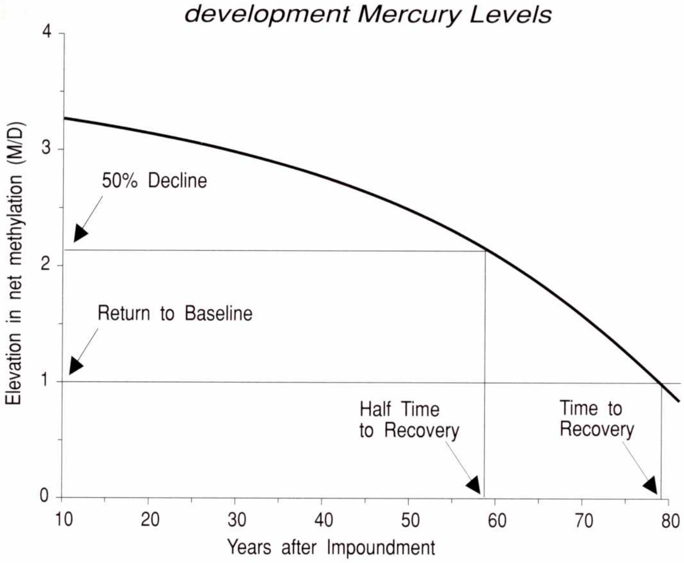
To determine if the process of net mercury methylation could be regulated, studies were conducted to identify the groups of bacteria responsible for mercury methylation and mercury demethylation. Identifications were made on the basis of experiments with specific metabolic inhibitors.

To identify recent temporal trends in mercury levels in fish, mercury levels in six species from 5 regions of SIL and from Issett Lake were measured. This mercury data extended the historical record for whitefish, which began in 1975, and for walleye and northern pike, which began in 1978 or 1979, depending on the site.

Duration of Elevated Mercury Levels

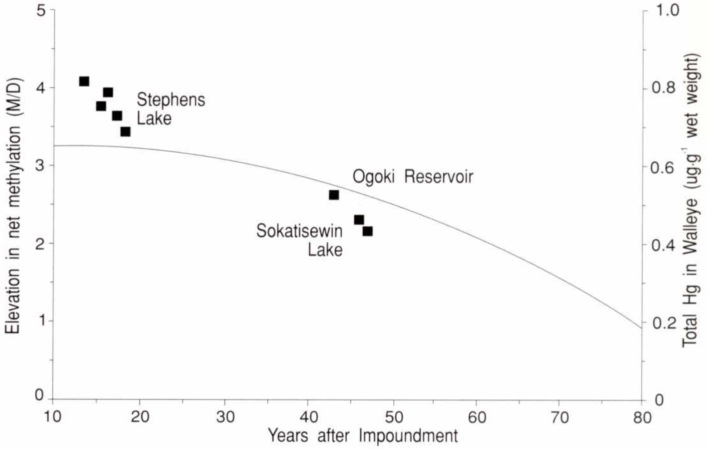
A model to predict the estimated time for a return to pre-development mercury levels was developed on the basis of the measured net rates of methyl mercury production in reservoirs of various ages (Fig. 2.14). (Because there are no pre-development data on net rates of methyl mercury production for any reservoir in northern Manitoba, Granville Lake data were used as a surrogate for pre-development conditions.) 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) is estimated to be 59 years, with an estimated return to pre-development levels 78 years after impoundment.

Figure 2.14 Estimated Time for a Return to Pre-development Mercury Levels



The validity of this model was evaluated by overlaying pike and walleye mercury data from selected reservoirs on the predicted decline curve. All fish mercury data were from reservoirs with the following similar characteristics: 1) similar extent of flooding; 2) location within the boreal forest zone; 3) the same major fish species; 4) comparable regional or pre-development baseline mercury levels; and 5) the absence of an upstream reservoir within 130 km. The only Canadian reservoirs which appeared to satisfy these requirements were Stephens and Sokatisewin lakes, and the Ogoki reservoir in north-western Ontario (flooded in 1943). As seen in Fig. 2.15, there is good agreement between the predicted net methyl mercury production and the measured mercury concentrations in fish.

Figure 2.15 Relationship between Elevation in Methylation Balance (M/D) and Mean Mercury Concentration in Walleye



The single most important factor affecting net methyl mercury production, and thus ultimately the duration of the mercury problem, is the amount and type of flooded organic material. This conclusion of the CMMA was reinforced by the FEMP mercury program, as evidenced, for example, by the results of the sampling program in Methyl Bay, SIL, in 1986. Although higher, on average, than at the offshore station at Methyl Bay or the Granville Lake stations, considerable variation in methylation activity was evident among the 3 near shore (flooded) stations (Fig. 2.16). An examination of the site characteristics revealed the following differences: 1) repeated sampling at station #1, since 1982, had effectively removed the organic layer; 2) oxygen and nitrogen concentrations at station #2 were the most similar to those encountered at flooded sites in previous years; and 3) organic carbon concentrations at station #3, a new station, were very low. Thus, the variation in methylation activity at these stations appeared to be the result of the substantial differences in sediment organic content among the sites.

Further evidence of the importance of the amount of organic material was provided by the higher than expected methylation activity in Sipiwesk Lake in 1987 in comparison to the predictions of the mercury duration model. It is postulated that the reason for this apparent contradiction of the model is the manner in which this reservoir is regulated. Since the operation of the Jenpeg dam in 1976, Sipiwesk Lake levels have been generally low in summer and high in winter, the reverse situation of most hydroelectric reservoirs in Canada. One of the results of this regime is exposure of the drawdown zone

Figure 2.16 Methylation, Demethylation, and Methylation/Demethylation Rates for Methyl Bay, SIL and Granville Lake, July 2 - September 3, 1986

Station No.[1]	Total Carbon	Organic Carbon	M	D	M/D
	mg (g dry sediment)$^{-1}$		% added Hg, $g^{-1} \cdot h^{-1}$		
Methyl Bay					
1	108[2]	103[2]	0.0028	0.0898	0.0312
2	191	187	0.0124	0.1616	0.0769
3	46	40	0.0003	0.0627	0.0042
4	22	21	0.0002	0.0146	0.0161
Granville Lake					
9	32	30	0.0006	0.0259	0.0224
10	30	29	0.0008	0.0302	0.0257
11	35	34	0.0010	0.0358	0.0289
12	23	22	0.0006	0.0127	0.0474

[1] Station numbers 1, 2, 3 are flooded near shore; 9, 10, 11 are reference near shore; and 4 and 12 are reference offshore

[2] Includes 1 estimated value in the average. All values are time-weight means.

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, providing a continuous source of fresh organic matter. Thus, it is clear that the model's prediction of a decline in mercury is based on a decline over time of the availability of organic matter.

The type, as well as the amount, of flooded organic matter will also determine its availability over time, as was evidenced by the mercury problem in Cedar Lake, Manitoba. Mercury concentrations in pike and walleye in Cedar Lake declined 12 to 14 years after its impoundment by the Grand Rapids Generating Station to levels commonly found in northern Manitoba lakes remote from hydro development. The type of vegetation flooded here, predominately marsh macrophytes and sedges, decomposed more quickly than the boreal forest vegetation flooded in the reservoirs in the FEMP study area and contributed much less (estimated 90%) organic matter per unit area of flooded land.

Another factor which may affect the duration of the mercury problem is sediment deposition. There is some suggestion that this may have happened in Rat Lake. A small but significant decline in mercury in northern pike in Rat Lake between 1986 and 1989 was observed coincident with an apparent increase in shoreline erosion in Notigi Reservoir. The covering of organic matter with inorganic sediment may be effectively preventing the stimulation of microbial methylation; the long-term impact of eroding clays on the duration of the mercury problem in the CRD is unclear and remains to be established.

The downstream transport of mercury in water has been suggested as a possible factor in determining both the magnitude and duration of elevated mercury levels in fish. Recent improvements in methodologies enabled the direct measurement of total mercury and methyl mercury in water as part of the FEMP mercury program. (See volume 1, chapter 4.) Methyl mercury levels in water sampled at Thompson were in the same range as that sampled in Granville Lake, suggesting that methyl mercury produced in Notigi Reservoir does not travel as far as Thompson, 130 km downstream. Thus, it would appear that locations 130 km or more downstream of a reservoir in the FEMP study area would not have elevated mercury levels in water due to elevated rates of methyl mercury production in the reservoir.

Mitigation

The potential for regulating the net production of methyl mercury was investigated; regulation of this process would provide an effective mitigative measure for elevated mercury levels in fish. Based on the results of a 1987 FEMP pilot study, and of investigations elsewhere, two groups of bacteria, methanogens and sulphate reducers, were identified as the most likely important bacteria in the net production of methyl mercury. The importance of each group was subsequently determined by measuring the reduction in the specific methylation and demethylation rates caused by the addition of a specific inhibitor for each group.

It was found that the methanogens and sulphate reducers jointly accounted for the majority of seasonal mean methylation (66%) and demethylation (50%) in Granville Lake sediments (Fig. 2.17). The two groups were equal contributors to both processes. Conditions in the flooded zone of Methyl Bay appear to be particularly attractive to those groups, where they jointly accounted for all of the methylation and demethylation. A group may have dominated one of the processes, but not the other, at a particular time and place, but both groups made equal contributions over the season on average.

Figure 2.17 Comparison of Seasonal Mean Specific Rates of Microbial Methylation and Demethylation Attributable to Methanogenic and Sulphate Reducing Bacteria, July 4 - August 24, 1988

% added Hg, $g^{-1} \cdot h^{-1}$

	Methylation			**Demethylation**		
Site	*Total*	*Methanogen*	*Sulphate Reducers*	*Total*	*Methanogen*	*Sulphate Reducers*
Methyl Bay, South Indian Lake						
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

All values are time-weighted means

The finding that the same bacterial groups are responsible for both methylation and demethylation indicates that it may be difficult to develop mitigative measures based on the selective inhibition of the methylators or stimulation of the demethylators. Other approaches, beyond the scope of FEMP, will have to be considered, such as an investigation of the factors regulating the transmission of the gene responsible for producing the demethylation enzyme within the bacterial community.

Microbial mercury methylation and demethylation rates were measured in sediments from a number of lakes and reservoirs representing a range of physical and chemical conditions to determine the most important factors 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 FEMP area is not limited by inorganic mercury availability.

The factor(s) that regulate microbial demethylation rates are less well understood, although organic matter and total microbial activity appear to be the most important. Demethylation rates were positively correlated with both sediment organic content and total microbial activity. Sediment pH, total mercury concentration, and oxygen availability had no significant influence on demethylation rates. This third factor 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.

It is currently not possible to mitigate the mercury problem through the regulation of the methylation and demethylation processes, either directly through the regulation of the methylating and demethylating bacteria or indirectly through the alteration of environmental conditions. Other alterations of environmental conditions, such as liming, selenium additions, and sediment resuspension, are similarly inappropriate as mitigation measures in the FEMP study area. Liming, a proposed solution to mercury problems related to lake acidification, would be ineffective in the FEMP study area reservoirs because the mercury problem here was not caused by a pH reduction. Selenium additions, while a demonstrated effective measure in reducing mercury bioaccumulation in aquatic food webs, must be made at concentrations ($1\text{-}5 \mu g \cdot L^{-1}$) that come uncomfortably close to the Canadian limit for drinking water ($10 \mu g \cdot L^{-1}$). Sediment resuspension as a remedial measure was tested under the CMMA; results showed that increased suspended sediment concentrations did not significantly reduce mercury concentrations in fish and caused slight increases in some cases. Indeed, the mercury problem developed at most locations in spite of increased suspended sediment concentrations.

Increasing fish growth has been suggested as a way of mitigating mercury problems; this suggestion is based on the hypothesis that if fish growth can be increased without causing a proportionate increase in methyl mercury availability, then the mercury concentration in fish would be reduced. Increased fish growth could be encouraged by intensive fishing and/or nutrient addition. However, neither of these methods may be effective in the FEMP study area. Intensive fishing is subject to some practical limitations, such as: 1) the need to have small enough lakes to make it feasible to significantly reduce fish standing stocks; 2) the potential for immigration of contaminated fish from other parts of the system must be limited; and 3) any trophy sport-fishing on a lake would be lost. These restrictions would limit the use of intensive fishing on most of the lakes in the CRD. Nutrient addition experiments were tried under the CMMA; while primary productivity increased, no effect on fish growth was noted. Furthermore, primary productivity in the turbid lakes of the CRD is generally light limited, so nutrient additions will not have a significant impact on primary productivity or fish growth.

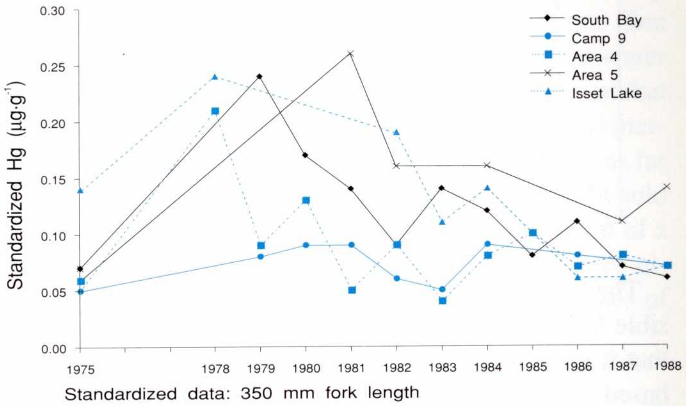
Thus, the only effective mercury mitigation measure continues to be the selective harvesting of fish with lower mercury levels, as suggested under the CMMA. This measure is based on the knowledge that some fish species such as whitefish and smaller-sized fish in a given species will have lower mercury concentration than predatory or larger-sized fish. The potential negative impact on future spawning stocks and the socioeconomic impact of selective harvesting on the resource users, especially the native communities in the FEMP study area, must be considered in the implementation of this measure.

Mercury Levels in Fish

SIL and Issett Lake have the longest data record for mercury levels in fish in reservoirs in the world. These records were extended by FEMP studies conducted in 1987 and 1988 in 4 regions of SIL, South Bay (Area 6), Camp 9 (Area 2), Area 4, and Area 5, as well as in Issett Lake. The objective of these studies was to examine trends and site-to-site differences that occurred in mercury concentrations in fish after more than 10 years since impoundment.

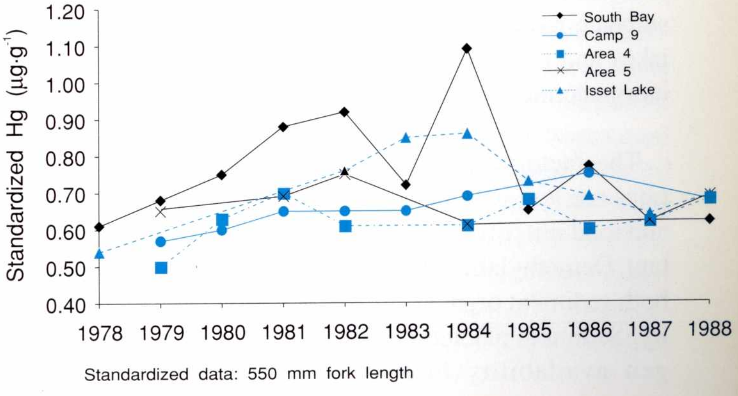
For most sites, mean muscle mercury concentrations in whitefish peaked in the late 1970s and have declined steadily to near 1975 values (Fig. 2.18). Area 5 whitefish, however, had mercury concentrations in 1988, 12 years after impoundment, which were double the concentrations in fish before impoundment. These Area 5 whitefish were slightly higher in mercury than those from Issett Lake, where water levels rose by more than 7 m causing more extensive flooding than in SIL.

Figure 2.18 Standardized Mean Muscle Mercury Concentrations for Whitefish in Southern Indian and Issett Lakes, 1975 - 1988



Mean muscle mercury concentrations for pike, at all sites, rose significantly immediately after flooding (Fig. 2.19). Mean values varied among sites and years but most means have been in the range of 0.5-1.0 µg·g^{-1} over the period 1980-1988. Highest values in 1988 and 1987 were in pike from Issett Lake; pike from Area 5 and South Bay were slightly lower in mercury than those from Issett Lake.

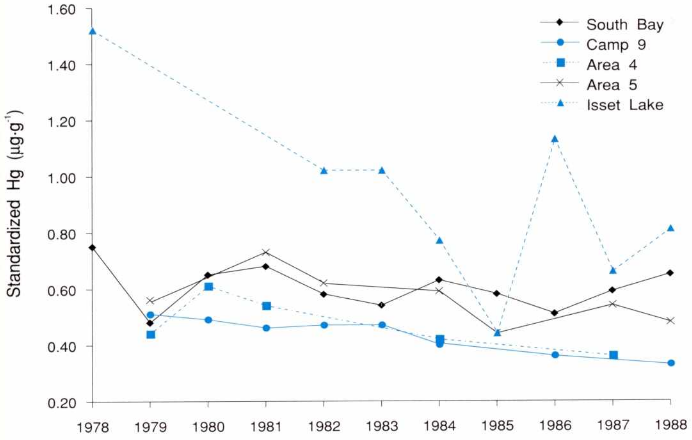
Figure 2.19 Standardized Mean Muscle Mercury Concentrations for Northern Pike in Southern Indian and Issett Lakes, 1978 - 1988



There was no indication of declining mercury concentrations in pike 12 years after impoundment of SIL.

Mean muscle mercury concentrations for walleye were, in general, much higher than lake whitefish and usually slightly lower than pike (Fig. 2.20). Means rose immediately after flooding, reaching values of $0.49 \mu g \cdot g^{-1}$ to $0.73 \mu g \cdot g^{-1}$ at most sites in SIL. In Issett Lake, the highest mean mercury concentration in walleye was recorded in 1978 at a value of $1.52 \mu g \cdot g^{-1}$. Mercury levels in walleye appear to have declined at some sites, but not others; however, year-to-year variability of means remains high, especially for fish from Issett Lake.

Figure 2.20 Standardized Mean Muscle Mercury Concentrations for Walleye in Southern Indian and Issett Lakes, 1978 - 1988



Standardized data: 400 mm fork length

The FEMP sampling results reaffirmed the conclusion of the CMMA that mercury concentrations in fish varied according to species, length of time since flooding, and location. Particularly notable was the large variation in mercury concentrations found even 12 years after flooding (Fig. 2.21). The year-to-year variability in mercury levels may be attributable, at least in part, to variability in the size distribution of fish in each year's sample. Means which were standardized or adjusted for length were not as variable year-to-year as arithmetic means, illustrating the effect of size on mercury concentration.

Figure 2.21 Ranges of Muscle Mercury Concentrations in Southern Indian and Issett Lakes, 1987 and 1988

Site	*Year*	*Muscle[Hg] Range ($\mu g \cdot g^{-1}$)*
Whitefish		
South Bay	1987	0.01 - 0.36
	1988	0.01 - 0.22
Camp 9	1988	0.01 - 0.34
Area 4	1987	0.03 - 0.24
	1988	0.04 - 0.14
Area 5	1987	0.06 - 0.55
	1988	0.08 - 0.39
Issett Lake	1987	0.02 - 0.52
	1988	0.02 - 0.52
Pike		
South Bay	1987	0.08 - 1.27
	1988	0.33 - 1.35
Camp 9	1988	0.29 - 1.14
Area 4	1987	0.25 - 0.99
	1988	0.37 - 1.04
Area 5	1987	0.36 - 0.99
	1988	0.45 - 1.93
Issett Lake	1987	0.28 - 1.84
	1988	0.15 - 2.67
Walleye		
South Bay	1987	0.38 - 1.28
	1988	0.27 - 2.88
Camp 9	1988	0.20 - 0.59
Area 4	1987	0.26 - 0.43
	1988	0.41 - 0.58*
Area 5	1987	0.24 - 1.04
	1988	0.28 - 1.13
Issett Lake	1987	0.33 - 1.97
	1988	0.19 - 2.23

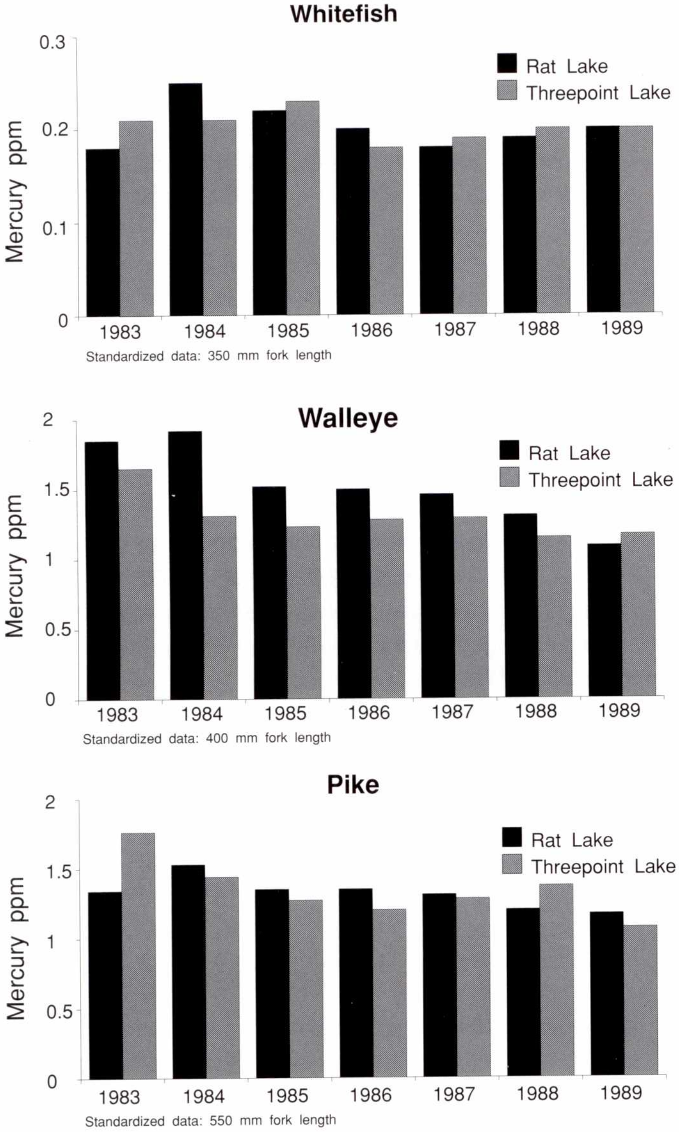
* only 2 samples collected

Three sites, Issett Lake, Area 5, and South Bay have been consistently higher in fish mercury concentrations than other sites in SIL. Issett Lake and area 5 of SIL were both extensively flooded; this similarity helps explain the similarity in fish mercury concentrations. It is more difficult to explain the occurrence of high mercury levels in South Bay. It may be that local conditions in South Bay have facilitated higher methylation activity here (e.g. through the introduction of organic material from the high erosion rates in South Bay) or that fish high in mercury are moving into South Bay from Issett Lake through the diversion channel.

Mercury concentrations in fish from Rat and Threepoint lakes were sampled by the Manitoba Department of

Natural Resources (DNR) from 1983 to 1989. While not part of FEMP, these results are reported here (Fig. 2.22) to help the reader to better understand the mercury problem in the CRD area.

Figure 2.22 Standardized Mean Muscle Mercury Concentrations for Fish in Rat and Threepoint Lakes, 1983 - 1989



Whitefish mercury levels in Rat Lake were stable between 1986 and 1989 at 0.2 $\mu g \cdot g^{-1}$; these levels are twice as high as the average for northern Manitoba lakes remote from hydro development (<0.1 $\mu g \cdot g^{-1}$) and 2.5 to 4 times the pre-development levels upstream in SIL (0.05-0.07 $\mu g \cdot g^{-1}$) and downstream in Wuskwatim Lake (0.08 $\mu g \cdot g^{-1}$ in 1970). Mercury levels in both walleye and pike declined over the DNR sampling period; however by 1989 levels were still about 3 times higher than the average for northern Manitoba lakes remote from hydro development (0.3-0.4 $\mu g \cdot g^{-1}$) and between 3 and 4 times the pre-development level upstream in SIL (walleye: 0.19 - 0.30$\mu g \cdot g^{-1}$ and pike: 0.26 - 0.32 $\mu g \cdot g^{-1}$).

Mercury levels in whitefish in Threepoint Lake, as in Rat Lake, were constant at about 0.2 $\mu g \cdot g^{-1}$, between 1983 and 1989. Mercury levels in both walleye and pike, during this period, were highest in 1983 at values of 1.65 and 1.76 respectively, followed by a noticeable decline in 1984. Further smaller declines were recorded between 1984 and 1989; however, 1989 values were still 3 to 4 times higher than northern Manitoba lakes remote from hydro development or pre-development levels upstream in SIL.

Mercury Levels in People

While not part of FEMP, the results of Health and Welfare Canada's (HWC) mercury testing program in South Indian Lake and Nelson House are reported here (Fig. 2.23) to provide the reader with a complete presentation of the available mercury information for the CRD area. Levels of 0 - 19 parts per billion (ppb) are considered to be the normal range for mercury for adults. This value was established by setting a safety factor of one-tenth of the level (200 to 500 ppb) at which signs and symptoms of methyl mercury poisoning would appear in the most sensitive adults. Because prenatal effects may occur at lower levels, women of child-bearing age have been an important target group for sampling by HWC.

Approximately 52% of the population of South Indian Lake was tested for mercury in 1987-1988. Of the people tested, approximately 80% had values in the range of 0-19 ppb; almost all of the remaining values were in the range of 20-50 ppb. For women of child-bearing age, almost 90% had values in the 0-19 ppb range.

Approximately 46% of the population of Nelson House was tested for mercury in 1987-1988. Of the people tested, approximately 90% had values in the range of 0-19 ppb; almost all of the remaining values were in the range of 20-50 ppb. For women of child-bearing age, approximately 98% were in the range of 0-19 ppb.

Figure 2.23 Mercury Levels in Residents of Nelson House and South Indian Lake

Mercury Levels in Residents of Nelson House

Year	*Sample Number*	*0 - 19 ppb*	*20 - 99 ppb*	*>99 ppb*
1976	4	1	3	0
1977	59	57	2	0
1978	102	63	38	1
1979	130	71	59	0
1980	0	0	0	0
1981	0	0	0	0
1982	56	50	6	0
1983	24	24	0	0
1984	221	181	39	1
1985	121	93	28	0
1987-88	609	540	69	0

1987-88 Groups	*Sample Number*	*0 - 19 ppb*	*20 - 50 ppb*	*50 - 80 ppb*	*80 - 100 ppb*
Men	268	234	33	1	0
Women:					
12-45 yrs	185	182	2	1	0
over 45 yrs	73	43	29	1	0
Children:					
1-4 yrs	10	10	0	0	0
5-12 yrs	73	71	2	0	0

Mercury Levels in Residents of South Indian Lake

Year	*Sample Number*	*0 - 19 ppb*	*20 - 99 ppb*	*>99 ppb*
1977	37	29	8	0
1978	12	9	3	0
1979	100	26	68	6
1980	1	1	0	0
1981	165	102	63	0
1982	389	274	115	0
1983	59	55	4	0
1984	105	70	35	0
1985	56	31	25	0
1987-88	344	274	69	1

1987-88 Groups	*Sample Number*	*0 - 19 ppb*	*20 - 50 ppb*	*50 - 80 ppb*	*80 - 100 ppb*
Men	73	44	28	0	1
Women:					
13-45 yrs	116	104	11	1	0
over 45 yrs	26	13	12	1	0
Children:					
1-4 yrs	77	72	5	0	0
5-12 yrs	52	41	11	0	0

HWC has been regularly testing mercury levels in the residents of all six native communities in the FEMP study area, since the mid 1970s, as part of their national testing program. However, a number of factors make it difficult to assess trends in mercury levels: 1) large variability in the number of community residents tested each year; 2) no identification in the published data record of values from the repeated testing of the same individuals; and 3) annual reporting of test results, which obscures any seasonal patterns.

Future Mercury Studies Along the Churchill River Diversion

The FEMP mercury studies identified several issues relevant to the CRD area that warrant further investigation, as follows: 1) factors affecting microbial demethylation; 2) downstream transport of mercury; 3) the potential of intensive fishing as a means of reducing mercury levels in fish; 4) the complete or partial removal of organic material as a means of preventing or minimizing mercury levels in fish; and 5) a case study of the area of the proposed Wuskwatim Generating Station.

Studies to date have suggested that microbial demethylation rates, like microbial methylation rates, are regulated by organic matter availability and total microbial activity; however, these observations need to be confirmed. There is also 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 methylating bacteria in flooded reservoir sediments.

FEMP results showed that any methyl mercury in water leaving the Notigi Reservoir is lost before Thompson; the mechanism by which any methyl mercury is lost from the water column in this reach should 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. In addition, the relative importance of fish and water as the mechanism(s) for downstream transport of mercury remains to be identified.