An investigation of the benefits, costs and long-term impacts on fish resources of intensive fishing as a potential means of reducing mercury levels in fish should be considered. A pilot project could be conducted on those off-route lakes (lakes off of the mainstream of the CRD), with naturally high mercury levels. These lakes are generally smaller and better isolated than those in the CRD making them more suitable for manipulation. Impacts on sport fisheries would also be more easily avoided. Because of these characteristics, these lakes might be a better site to test the effectiveness of intensive fishing than those lakes on the CRD.

Another pilot project which could be undertaken is an evaluation of the effectiveness and an assessment of adverse environmental side effects of the controlled burning of organic matter as a measure to minimize or prevent elevated fish mercury levels in new reservoirs. Preliminary investigations could employ limnocorral experiments to test the ability of various terrestrial organic materials to stimulate microbial methylation and the duration of this stimulation. Field trials in a new reservoir could then be conducted if the initial results of the limnocorral experiments appear favourable.

The reservoir to be created by the proposed Wuskwatim Dam on the Burntwood River would provide an ideal site for the controlled burn pilot project, as well as an opportunity to apply and expand the knowledge of mercury in the CRD generated by the results of the FEMP and CMMA studies. For example, measurements of methyl mercury in water and of methylation and demethylation rates could be taken at the reservoir site and immediately downstream, before the reservoir is created, during the construction period, and immediately after the formation of the reservoir. These data, the first such "pre-" and "early development" data anywhere in the FEMP study area, would be invaluable in refining the mercury duration model. (These data are considered "pre-" and "early development" with respect to the construction of Wuskwatim Dam only. Wuskwatim Lake is not a "pristine" lake; it has already been flooded as a result of the increased flows down the Churchill River Diversion.) In addition, these data if supplemented by data on mercury in fish from the proposed reservoir site and immediately downstream, would provide new information on the understanding of the downstream transport of mercury.

PIKE POPULATIONS IN SOUTHERN INDIAN LAKE

Pike is important to the commercial fishery at Southern Indian Lake: 1) it is the final host in the life-cycle of the parasite *Triaenophorus crassus*, a parasite which lowers the market quality of lake whitefish, the most economically important species of fish harvested in SIL; and 2) it can be an important source of income when whitefish prices are low.

In 1976, DFO initiated a study of pike population in Wupaw Bay SIL, an isolated body of water largely unaffected by the flow of the Churchill River. Although not completely typical of much of post-impoundment SIL because of its short wind fetches and small flooded terrestrial area, this bay offered a sheltered workable area with a good population of pike. The main finding of this 6-year study was a significantly higher pike spawning success in 1977 only, the first year following flooding. It was hypothesized that the flooded terrestrial vegetation immediately following impoundment provided pike with an abundance of substrate on which to lay their eggs and probably also provided a favourable nursery area for the pike fry. A highly successful spawn, combined with an abundance of preferred food and protection from predators were responsible for a highly successful pike year class in Wupaw Bay in 1977. The decline in subsequent year classes was attributed to physical changes in the littoral zone, especially the degradation of the submerged vegetation.

Under the auspices of FEMP, the study of pike populations in Wupaw Bay was extended to 1988 to further examine the effects of impoundment of SIL on the pike and on the forage fish that constitute part of the pike's diet. Throughout the summers of 1982-84 and 1986-88, gill nets were set at sampling sites chosen at random from a total of 36 sites distributed around Wupaw Bay. Length, weight, sex, age, condition factor (a measure of fitness), and stomach contents were recorded for pike captured from gill nets. Once or twice a week each summer five seining stations were sampled by pulling seine nets by hand. The number of fish of each species caught were recorded, identified, and measured.

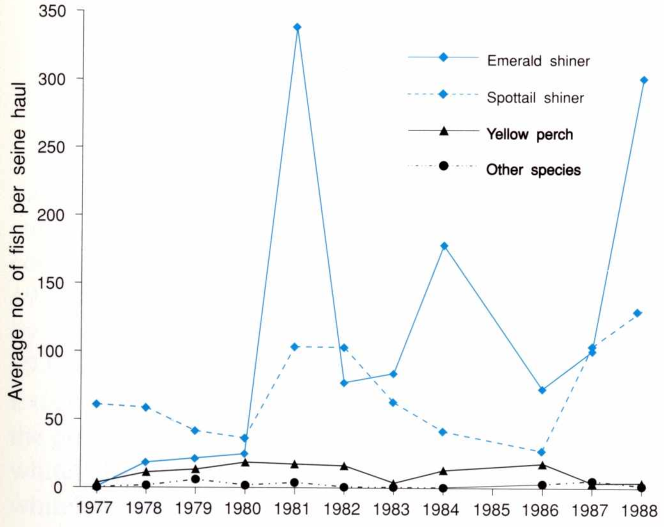
Analysis of the entire data record, from 1976 to 1988, suggest that the adult pike population in Wupaw Bay has suffered since lake impoundment. A downward trend was observed in both catch per unit of effort (from 13.52

per 8 hour period in 1976 to 5.60 in 1988) and in condition factor (from 0.89 in 1976 to 0.84 in 1988). These decreases reflected the status of the entire population; they were not the result of any dominance by the 1977 year class.

The decline in pike abundance appears to be related to physical changes in Wupaw Bay such as an increase in depth (from a range of 0 to 3 m to 0 to 6 m) and a decrease in areas of macrophytes. Before flooding, much of the bay was suitable pike habitat, but since flooding, favoured pike habitat has been restricted to a few peripheral areas where gently sloped shorelines were extensively flooded. A habitat well suited for pike was transformed into one that, at least in relative terms, favoured open-water species. The results of the forage fish study reaffirmed this conclusion.

Shiners (emeralds plus spottails) dominated the seine catches every year, comprising more than 90% of the catch in 7 of the 11 years of seining data. Beginning in 1981, the number of emeralds, relative to the number of spottails, increased noticeably (Fig. 2.24). It is hypothesized that the creation of a deeper clearer body of water in Wupaw Bay (in contrast to the larger basins of SIL) provided a more favourable habitat for the pelagic emerald shiner relative to the more littoral species such as the spottail shiner. It is likely that this stimulatory effect on emerald shiner abundance was limited to Wupaw Bay and similar shallow isolated bays where the post-impoundment increase in mean depth was significant.

Figure 2.24 Average Catch per Seine Haul in Wupaw Bay, Southern Indian Lake, 1977 - 1988



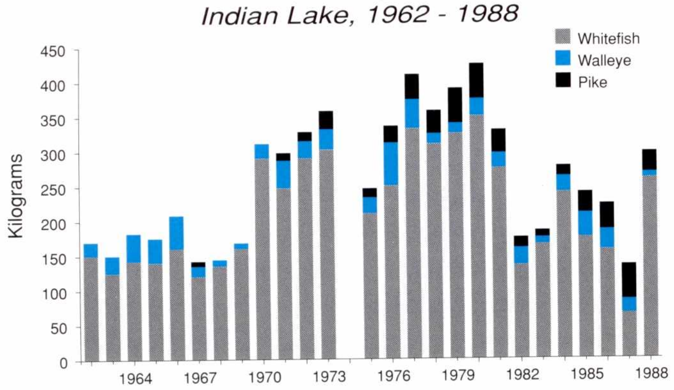
The abundance of littoral species, such as spottail shiner and yellow perch, fluctuated over the study period but showed no overall trend. Spottail shiners were much more abundant in 1982 and 1988 due to high over-winter survival rates of the 1981 and 1987 year-classes respectively, when temperatures were warmer than average. Warmer water temperatures generally result in faster growth of fish, which enhances their ability to avoid predators. Thus, it appears that weather over the open-water season had a greater impact on the abundance of spottail shiners and yellow perch in Wupaw Bay than did the flooding of the lake.

The deterioration in condition of Wupaw Bay pike is thus puzzling, considering the apparently abundant food supply (e.g. shiners, perch) after impoundment of SIL. The percentage of pike stomachs that were empty declined immediately after flooding (from 76% in 1976 to 50% in 1978) then increased slowly to near 1976 levels, by 1988. There was considerable variability over time in the contents of the pike stomachs; for example, the importance of shiners in the pike diet declined after 1977, with the large number of shiners found in stomachs in 1986 being an anomaly to the overall trend. Yellow perch were found in only 4% of stomachs in 1977, increased to 12% in 1981, and afterwards their occurrence in stomachs declined. There seemed to be no relationship between large year classes or overall abundance of forage fish and their occurrence in pike diets.

BIOLOGICAL ASPECTS OF THE WHITEFISH COMMERCIAL FISHERY AT SOUTHERN INDIAN LAKE

The commercial fishery at SIL was the largest in northern Manitoba prior to impoundment and diversion with about 333,500 kg of fish taken yearly. Approximately 85% of the total commercial catch weight was composed of lake whitefish, with the rest being made up of pike and walleye (Fig. 2.25). Immediately following impoundment there was a substantial drop in catch per unit effort (CPU) (Fig. 2.26). In 1982, the whole whitefish catch of the lake was downgraded from export to continental grade, with a concomitant substantial drop in fish price. The lower price and reduced CPU led directly to a collapse of the commercial fishery.

Figure 2.25 Yearly Catch of the Three Main Commercial Species at Southern Indian Lake, 1962 - 1988



Under the auspices of FEMP, a biological assessment of the post-impoundment commercial fishery at SIL was undertaken, with quantitative data collected for the 1987 open-water season and qualitative data only for the 1988 season. Data were obtained through interviews with commercial fishermen, from field studies, and from a review of the catch records from the Freshwater Fish Marketing Corporation (FFMC), a Crown corporation owned by the federal government which holds a virtual monopoly on the sale of freshwater fish in parts of Canada, including Manitoba. The FEMP sponsored study focused on the fish delivered to the Missi Falls plant since they comprised the majority of the SIL commercial catch. Information was collected on whitefish numbers, length, weight, age, colour, grade, cyst count, CPU, catch location, and seasonal analysis of catch.

The 1987 data showed that CPU was still depressed in comparison to pre-impoundment levels, both for Region 4, historically the most important commercial fishing area of SIL, and for Regions 4 and 5 combined, where more of the post-impoundment fishing activity is occurring (Fig. 2.26). The number of days nets are left in one location has been reduced from a pre-impoundment average of 18 days to an average in 1987 of 6.3 days in Region 4 and 5.0 days in Region 5. Frequent relocation of nets (e.g. due to fouling by debris and/or non-commercially valuable rough fish) reduces the amount of time that nets are in place for catching fish, and increases fishing effort. However, both CPU and number of days nets left in one location were higher in 1987 than in 1981. (The most important factor affecting CPU, the abundance of fish and their movement, is discussed in the next section.)

The 1987 open water season had one of the lowest total catch weights of the last 25 commercial open water seasons (140,000 kg) and one of the lowest whitefish percentage of catch (46%) (Fig. 2.25). Pike comprised an unusually high proportion (42%) of the total 1987 commercial catch. The 1988 fishery was more comparable to the pre-impoundment fishery in total catch weight (300,000 kg) and species composition (whitefish percentage of catch was 85%).

A major factor for the differences between the 1987 and 1988 open-water seasons was the higher whitefish prices in 1988. The low whitefish prices in 1987 promoted a dichotomy in the fishery, with pike being the favoured species in the spring (comprising approximately 61% of the catch) and whitefish being the favoured species in the fall season (comprising approximately 93% of the catch). Greater movements of pike in

Figure 2.26 Selected Statistics for the Open Water Commercial Fishery at Southern Indian Lake

	1987 R.4	*1987 R.4/5*	*1981*	*1980*	*1979*	*1972*
Catch per unit effort (kg/n.n)	8.44	9.75	7.50	10.30	15.50	23.10
% Whitefish export grade	37.00	33.00	19.00	66.00	28.00	100.00
Mean cyst count per kg dressed	1.05	1.20	1.16	1.09	1.22	0.53
Mean cyst count per shipment	51.90	61.40	57.20	54.20	79.10	23.10
% light colored whitefish	66.00	59.00	14.00	65.00	28.00	100.00
% Whitefish in commercial catch*	71.00	46.00	79.00	89.00	81.00	91.00
% Total effort	58.00		67.00	70.00	38.00	100.00
Days net left at one location	6.30	5.00	5.60	5.00	7.00	18.00**
Whitefish mean age***	9.80	9.70	10.20	10.40	9.95	9.10

* Data from FFMC, except 1987, Region 4, from FEMP study
** Pre flood average
*** From fin ray sections except 1972, from scale analysis

the spring and whitefish in the fall, due to spawning activities, make them easier to catch at those times of the year. The shift in effort between species appeared to reflect an opportunistic attempt by the fishermen to continue fishing by exploiting those species with the best chance of large economic returns. (Another reason for the higher catches in 1988 would have been the introduction of the 12.7 cm mesh nets into the whitefish fishery; the slightly smaller mesh size would have increased the catch because whitefish should have been more vulnerable to being caught in the smaller mesh.)

Fish prices are extremely important to a commercial fishery in a northern lake, like SIL, that is remote from southern markets. Whitefish prices can vary substantially depending on the grade - export, continental, or cutter (Fig. 2.27). Grade classifications are based on the number of *Triaenophorus crassus* cysts present per 45 kg of whitefish (dressed weight): export grade, 0 to 40 cysts per 45 kg; continental grade, 41 to 80 cysts per 45 kg; and cutter grade, over 80 cysts per 45 kg.

Figure 2.27 Fresh Whitefish Prices, Summer 1988, FOB Station - Leaf Rapids

	Grade	$./kg
Whitefish	sml (.45-.7)	0.412
Export	med (.7-1.4)	0.792
Dressed	lge (1.4-1.8)	0.812
	Jmbo (over 1.8)	0.832
Whitefish	sml (.45-.7)	0.302
Continental	med (.7-1.4)	0.412
Dressed	lge (1.4-1.8)	0.412
	Jmbo (over 1.8)	0.412
Cutters/Hdls	all sizes	0.372

The average cyst count for fish sampled in 1987 under the FEMP study were: Region 4, 51.9; Region 5, 74.1; and the southern portion of the lake that delivers to the community, 81.1. Data on the cyst counts in whitefish in 1987 were further analyzed by sub-areas of Region 4, and by colour of whitefish. The cyst count for Region 4A was 59.9, Region 4B was 21.3, and Region 4C was 64.1. Except for Region 4B, where the sample size was small, the grade was still continental. Analyses of the sampled whitefish by colour showed that 65% of the export whitefish were light coloured and that the chance of a light whitefish being export was 48%. Thus, it appears that neither selective fishing by sub-area of Region 4, the traditional export grade area of SIL, nor by colour for light whitefish, would likely result in a reclassification of the commercial catch from continental to export. However, it should be noted that in 1987 Region 4 still produced the highest quality fish of the sampled regions.

The higher cyst counts found in the whitefish from Region 5, in 1987, agreed with historical evidence that Region 5 produced poorer quality whitefish than Region 4. Previous research (Johnston 1984) found that conditions favourable for higher cyst levels are more prevalent in Region 5 than in Region 4, due to the shallower mean water depth and the larger numbers of pike, the final host in the parasite cyst cycle. Bodaly et al. (1980) indicated that decreased light penetration after flooding could have caused whitefish to spend more time in shallow water. This would bring them into a situation where there is an increased probability of cyst infestation, as a result of co-inhabiting with pike where pike are more prevalent. Another factor which may result in higher average cyst counts in the commercial catch is the harvesting of smaller whitefish, through the use of nets with smaller mesh sizes, since larger whitefish can have lower rates of cyst infestation.

Age analysis of the 1987 sampled whitefish showed that the majority of individuals were between 8 and 11 years; the mean age (9.72) was quite similar to that found from 1979 to 1981 (9.95 - 10.40). There was very little difference between Regions 4 and 5 in the overall age structure of the sampled catch. Earlier studies (Ayles 1976) found similar mean ages. The age distribution of the commercial catch suggests that SIL is a light to moderately exploited commercial whitefish fishery; this low level of commercial exploitation would render unlikely any hypothesis that would consider over fishing as a causal factor in the decreased CPU levels on traditional fishing grounds.

The results of the biological assessment of the SIL fishery suggest that: 1) the fishery may be stabilizing at levels lower than pre-impoundment values or that a marginal improvement may be taking place; 2) a return to an export quality fishery will not be an option, at least not for the immediate future; and 3) there was a shift in the geographical distribution of fishing effort from Region 4 to Region 5 in response to higher catches of preferred species.

WHITEFISH DOWNSTREAM OF MISSI FALLS

Bodaly et al. (1984) suggested that declines in catch per unit (CPU) effort in SIL may be due to major movements of fish out of the lake in response to changes caused by impoundment. Large numbers of whitefish have been observed congregating immediately downstream of the Missi Falls Control Structure on SIL. The objective of one FEMP-sponsored study was to estimate the numbers and seasonal pattern of lake whitefish abundance below Missi Falls and to determine the origin of these fish.

Whitefish were live sampled below Missi Falls, 5 times in 1986 and 4 times in 1987 during the open-water seasons. Fish in each sampling group were given a distinctive marking (e.g., dorsal fin clip) that distinguished them from the other sampling groups. Recapture of previously marked fish provided estimates of population numbers below the Missi Falls Control Structure. Samples were also obtained to provide morphological and biological comparisons with whitefish caught at 2 upstream locations, Region 4 and near Strawberry Island in SIL, and at 4 downstream locations, Partridge Breast, Northern Indian, Fidler and Gauer lakes. Morphological differences among whitefish are invaluable in indicating stock discreteness, especially in instances where genetic differences do not exist, as was the case for whitefish in the Missi Falls area (Bodaly et al. 1984). Biological data were used to compare the health and reproductive readiness of the sampled whitefish.

Large numbers of fish were caught below Missi Falls in both study years, with whitefish dominating the catch for nearly all sample times. Estimated whitefish populations ranged from a low of 10,701 in July 1987 to a high of 88,764 in September 1986. (Fig. 2.28). In addition to the significantly larger number of fish present below Missi Falls in 1986 than in 1987, there was also a difference in the seasonal distribution. In 1986, estimated population numbers gradually increased over the summer reaching a peak in the autumn; in 1987, however, the population numbers peaked early in the summer.

Examination of the morphological data set, without the Missi Falls samples, indicated that fish from the upstream and downstream areas were quite morphologically different from one another. Because of the long sampling period, from June 1986 to October 1987, there was a concern that temporal, rather than spatial, differences might have been responsible for the observed morphological differences between upstream and downstream sites. However, there was some evidence to suggest that morphological changes among locations did not represent regular seasonal changes within a single population. For example, if changes in morphology were largely due to seasonal changes, then the changes observed over time at Missi Falls in 1986 would have been reflected in the changes during the 1987 period; this was certainly not the case.

Figure 2.28 Estimated Whitefish Population at Missi Falls

Date	Number of Fish tagged	Estimated Pop Size	s.d.
Jun-86	355	-	-
Jul-86	598	13942	5744
Aug-86	610	44818	20528
Early Sep-86	2794	88764	21415
Late Sep-86	1577	88567	23042
Jun-87	3235	28867	5640
Jul-87	2529	10701	1625
Aug-87	1856	13454	3242
Sep-87	953	-	-

Comparison of the morphological data on the whitefish sampled at Missi Falls to the data on the whitefish sampled at the upstream and downstream sites showed, in general, that the Missi Falls fish caught in the autumn of 1986 most resembled the closest upstream samples, whereas the Missi Falls fish caught in the summer of 1987 most resembled the closest downstream samples. Thus, it was concluded that most of the whitefish at Missi Falls in 1986 appeared to have originated from upstream, in SIL, while in 1987 most of the Missi Falls whitefish appeared to have originated from the downstream lakes on the lower Churchill River.

Analysis of the biological data showed that the Missi Falls fish caught in 1987 were experiencing greater physiological stress than the 1986 fish sampled (e.g. the 1987 fish had significantly thinner bodies than the 1986 fish and had more empty digestive tracts) (Fig. 2.29). Gonad somatic indices (GSI), a measure of reproductive readiness, increased almost exponentially in 1986 for both males and females, but decreased over the 1987 sample season until autumn (Fig. 2.30).

Thus, there were marked differences in the characteristics of the whitefish populations below Missi Falls in 1986 and 1987. In 1986, a year of high discharge over Missi Falls: 1) there was a large number of whitefish present; 2) the numbers peaked in the autumn; 3) the fish were well-fed and in good condition; and 4) the majority appeared to have originated from SIL. It was concluded that they were trying to return to SIL to spawn. In 1987, a year of lower flows and higher water temperatures than 1986 (Fig. 2.31): 1) the number of whitefish present was lower than in 1986; 2) the numbers peaked in the summer; 3) the fish were in poorer condition than in 1986; and 4) the majority appeared to have originated from the downstream lakes. It was concluded that they were trying to escape the warmer waters of the downstream lakes.

It would appear that the reduction in CPU in SIL has been greatly influenced by the construction of the Missi Falls Control Structure. This study suggests that even 10 years after the impoundment of SIL, there are still significant numbers of lake whitefish attempting to complete the traditional migration. Future consideration should be given to mitigating measures which could involve a seining program at Missi Falls to transport fish upstream

Figure 2.29 Whitefish Digestive Tract Fullness, Missi Falls, 1986 and 1987

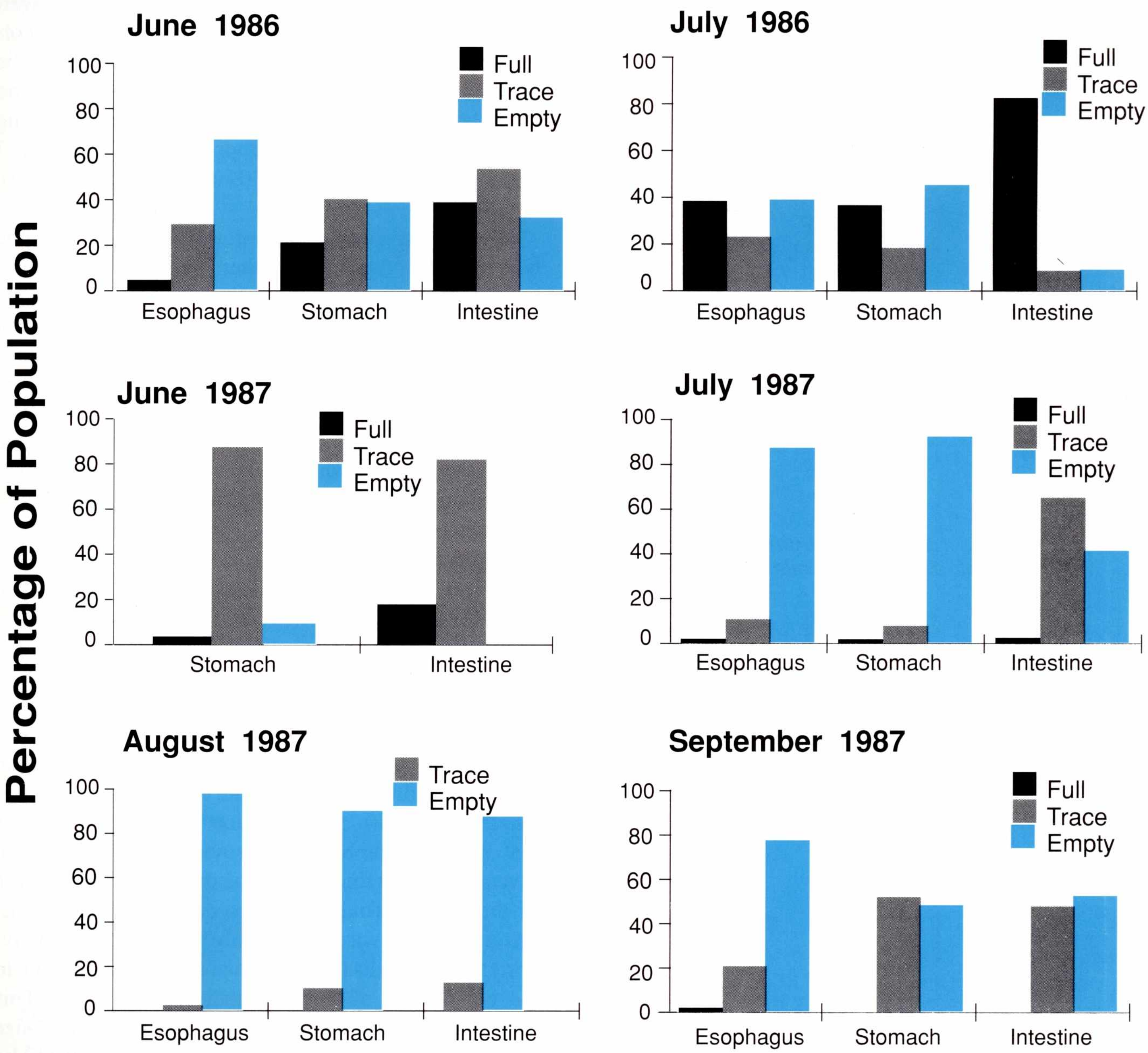
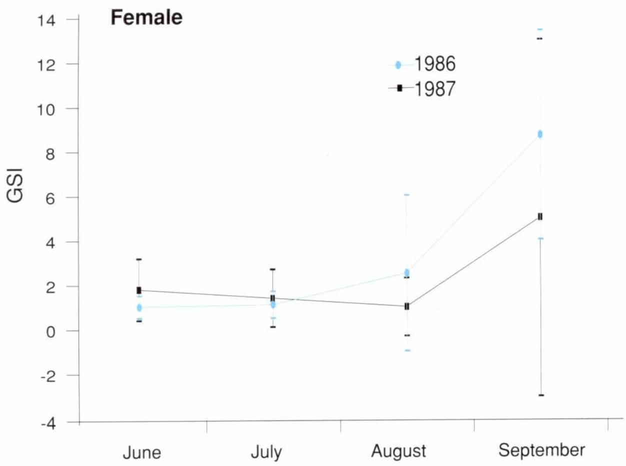


Figure 2.30 Temporal Changes in Gonad Somatic Index (GSI) for Whitefish at Missi Falls, 1986 and 1987



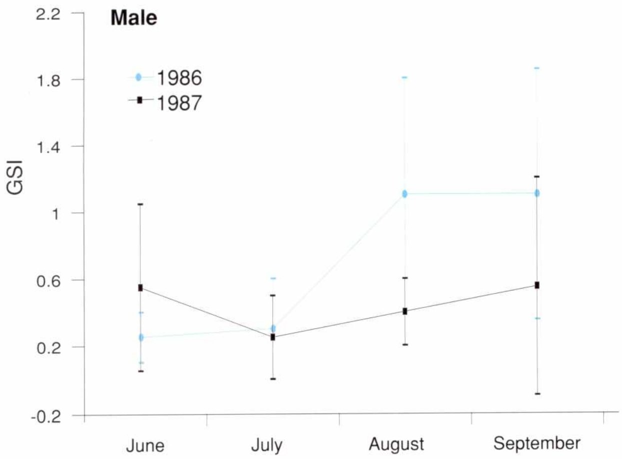
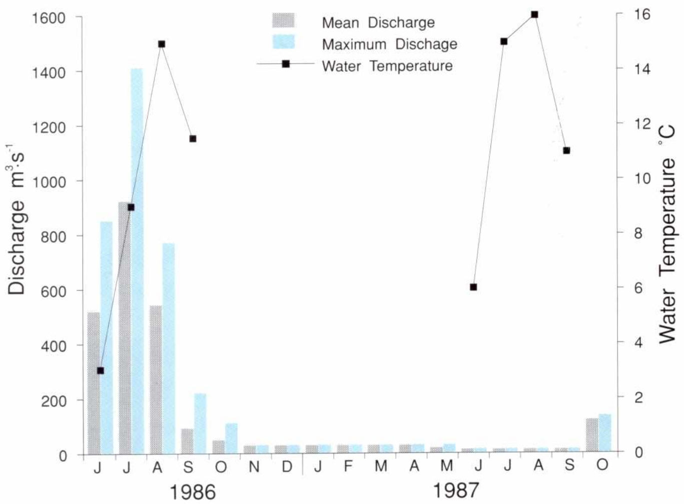


Figure 2.31 Discharge and Water Temperature Values at Missi Falls, 1986 and 1987



over the dam at critical periods when fish are attempting to move upstream.

THE ECONOMIC PERFORMANCE OF THE SOUTHERN INDIAN LAKE COMMERCIAL FISHERY

Since its inception in 1941, the commercial fishery at Southern Indian Lake has been a major component of the South Indian Lake economy. For example, it was estimated that in 1972, the last year before work began on raising the water level of SIL, the fishery was the single most important source of livelihood in the community, providing 43% of the community's income. However, following impoundment, the commercial fishery collapsed. Studies of the economic performance of the summer fishery at Southern Indian Lake in 1977 and 1980 showed that the net income per firm, without taking into account compensation, dropped from $1964 in 1977 (in 1980 dollars) to a loss of $304 in 1980 (Wagner 1984).

A FEMP sponsored study reassessed the economic health of the commercial fishery in 1988. Techniques used in this study included field research conducted between June 8, 1988 and September 5, 1988; participant observation, (e.g. accompanying fishermen onto the lake as an extra helper); conversations with fishermen, including the use of formal interviews with structured questionnaires; and a review of the data made available by the Freshwater Fish Marketing Corporation (FFMC). These techniques provided data on 23 firms, which represented 81% of the active fishermen on SIL. Fishing revenues and costs were calculated from 3 perspectives: 1) direct comparison to the 1980 study categories; 2) an approximation of the fishermen's view of the economics of the fishery; and 3) an estimation of the total community income from the fishery.

In 1988, the fishery produced an average net income for the fishermen of $8289 (Fig. 2.32). This calculation included an allowance for capital depreciation, and payments from hydro compensation and freight subsidies that in 1988 combined to provide 36% of the total revenue; without these supplements, the economic future of the fishery would be less secure. The average net income per firm was substantially higher for firms delivering to Missi Falls ($13754) than for firms delivering to the community of South Indian Lake ($4776). This difference was due primarily to the difference in the size of the average catch for firms in each region, 139,113 kg or 15,475 kg/firm, for the Missi Falls area and 81,596 kg or 5,828 kg/firm for the south end of the lake. There were also small differences between the two regions in average cost per kilogram of fish (the north end cost was \$0.80 versus \$0.89 in the south end) and in average revenue per kilogram of fish (the north end was \$1.69 versus \$1.72 in the south end). Differences in economy of scale for the larger catches in the north end and differences in species composition of the catches likely accounted for these small differences.

The average net income per firm increased from \$6,300 in 1980 (in 1988 dollars) to \$8,289 in 1988, an increase in real income of 32% (Fig. 2.32). The most important reason for this increase was a substantial increase in continental and cutter grade whitefish prices in 1988, the only grades now produced since the loss of the export grade rating for SIL in 1982 (Fig. 2.27).

Revenues from the SIL fishery since 1980 have also been affected by a change in the administration of the compensation fund. From 1978 to 1983, compensation to SIL fishermen varied depending on species, grades, and the area of the lake in which the fish were caught. Since 1983 a \$2.5 million compensation fund, administered by the South Indian Lake Fishermen's Association, pays

Figure 2.32 Aggregate and Average Net Incomes of Sampled Firms in Southern Indian Lake, 1980 and 1988

Items/Year	1988 SIL Deliveries	1988 Missi Falls Deliveries	1988 Aggregate	1980
Revenue				
Sales	\$ 91947	\$ 148337	\$ 240284	\$ 233150
Hydro Compensation	25356	47470	72826	68088
Freight Subsidies	23089	39792	62881	58803
Total	\$ 140392	\$ 235599	\$ 375991	\$ 360041
Costs				
Food & Fuel	27944	40906	68850	57369
Repairs	2231	2968	5199	8807
Fishing Gear	3023	2509	5532	18474
Hired Labour	1050	7610	8660	nil
Boat Charges	2255	24075	26330	41593
Truck Charges	10897	3126	14023	n/a
Ice Harvest	1768	3061	4829	n/a
Licenses	330	210	540	580
Insurance	nil	nil	nil	473
UIC	968	622	1590	2400
Miscellaneous	300	0	300	2178
Total	\$ 50766	\$ 85087	\$ 135853	\$ 131874
Depreciation				
Boats	4268	4150	8418	7412
Motors	11460	13760	25220	35970
Nets	5082	3798	8880	24333
Camp Gear	514	1025	1539	1436
Total	\$ 21324	\$ 22733	\$ 44057	\$ 69151
Interest	1431	3989	5420	7816
Aggregate Net Income	\$ 66871	\$ 123790	\$ 190661	\$ 151210
Average Net Income	\$ 4776	\$ 13754	\$ 8289	\$ 6300
	(14 firms)	(9 firms)	(23 firms)	(24 firms)

NOTE: All values are in 1988 \$, n/a = not applicable one amount for all commercial species from all areas of the lake and from certain outlying lakes. In 1988 this amount was $0.33/kg.

Another change that has occurred in the SIL fishery since the 1980 study was the introduction of a regular ferry service to the community in 1985. The daily transport of the fish catch to Leaf Rapids which regular ferry services enables, along with the south end's warmer temperatures, less rough waters, and close proximity to the community has led to the development of a community-based fishery in the south end of SIL, despite the smaller catches in comparison to the lake's north end. Greater commitment is required to fish in the less hospitable north end of the lake, with 10 to 12 hour days not uncommon for firms in this area in 1988. All of the fishermen in this area lived in camps located near where they were fishing, with trips to the community, a 2 hour boat ride, by these fishermen being relatively infrequent.

An analysis of the economics of the fishery from the fishermen's view differed from the preceding analysis in the following ways: 1) payments by the fishermen to the Manitoba Agricultural Credit Corporation (MACC) were used to represent the cost of capital items; 2) the cost of nets and camp gear purchased were used in place of the calculation of depreciation of these items; and 3) UIC benefits, as well as costs, were included. The resulting calculation, (Fig. 2.33) is significant because it more closely reflected the net cash income that was received by the fishermen. Using this form of analysis, the average net income per firm was $9,751.

An estimate of the total community income from the fishery was generated by calculating: 1) total estimated net income from the commercial fishery at SIL; 2) the value of the income-in-kind that resulted from the pursuit of the commercial fishery at SIL; 3) estimated net income for SIL fishermen from the inland lakes fishery; and 4) unemployment insurance benefits received by SIL fishermen. Total community income from the commercial fishery on SIL was estimated by assuming that the revenues and costs/kg of the firms not surveyed were similar to those firms which were surveyed. The aggregate revenues and costs of all firms were then calculated by using the ratio of the total number of kilograms of fish caught in SIL overall (267,908 kg) to the total caught by the firms surveyed (220,709 kg), i.e. 1.21:1. Multiplying the numbers in Fig. 2.34 by 1.21 gave an estimated total community income of $232,326. (In this calculation, UIC premiums were not included and the license cost was changed to $950, since 95 licenses were issued to the community.) If the wage labour paid out by the firms is added back in, the total estimated net income to the community in 1988 rises to $242,631.

Income-in-kind to the community that resulted from the pursuit of the commercial fishery activity at SIL was estimated as the value of domestic fish consumption and of moose shot while fishermen were on the lake fishing. Two estimates of domestic fish consumption were used: 1) an average consumption of 60 kg per person per year, based on studies done by various researchers in native communities across the north; and 2) an estimate of 30 kg per person per year, because of the ready availability of store bought goods in the community as well as in Leaf Rapids. Assuming a price of $4.00 per kg the value of the domestic fish consumed was estimated at $120,000 to $240,000. (It was observed that the fish consumed were commonly not of commercial value; i.e. they were either not a commercially valuable species, such as red sucker,

Figure 2.33 An Alternative Analysis of the Net Income of the Sampled Firms in Southern Indian Lake, 1988

Revenue	
Sales	$240,284.
Hydro Compensation	72,826.
Freight Subsidies	62,881.
UIC Benefits (estimated)	45,269.
Total Revenue	$421,260.
Costs	
Food and Fuel	68,850.
Repairs	5,199.
Fishing Gear	5,532.
Net Purchases	6,456.
Hired Labour	8,660.
Licenses	540.
Boat Charges	26,330.
Truck Charges	14,023.
Ice Harvest	4,829.
Camp Gear Purchases	465.
UIC Premiums	1,590.
Miscellaneous	300.
Total Operating Costs	142,774.
MACC Payments (incl interest)	54,216.
Aggregate Net Income	$224,270.
Average Net Income	$9,751.
	(23 firms.)

or they had been slightly damaged in the nets.) An estimated 20 moose were killed during the fishing season by SIL fishermen; most were opportunistic kills that would not have occurred had the people not been in the area fishing at the time. Each moose was valued at a minimum of $2,300 which reflected an estimated value of 250 kg of meat at $8.00 per kg and a value for the hide of $300.00. The total income-in-kind was thus estimated at $166,000 to $286,000.

In the 1980s, some SIL fishermen also began fishing nearby lakes, with financial assistance provided by the SIL Fishermen's Association to offset the higher transportation costs. Data collected in 1988 suggest that the inland lake fishery was more lucrative than the SIL fishery; the average income for 5 firms surveyed that operated on the inland lakes out of Missi Falls was $10,143 per firm. Access to the inland lakes fishery is limited, however, by the relative scarcity of lakes suitable for commercial fishing in the region, the limited availability of inland lake licenses, and the high start-up cost of operating on an inland lake.

Employment provided by the commercial fishery at SIL enabled 13 fishermen to meet eligibility requirements for unemployment insurance benefits, providing a net benefit to the community of $52,875. The limited availability of alternative forms of employment in SIL emphasizes the importance of this benefit.

Total community income from the commercial fishery in 1988 was estimated, roughly, at $512,221 to $632,221 consisting of: 1) net income from the SIL commercial fishery, including wages to hired labour, of $242, 631; 2) income-in-kind of $166,000 to $286,000; 3) net income for 5 SIL fishing firms from the inland lakes fishery of $50,715; and 4) unemployment benefits to SIL fishermen of $52,875. All of this income was directly dependant on the economic viability of the commercial fishery at SIL (e.g. provision of capital items, employment opportunities, etc.) Data collected on the summer commercial fishery in SIL in 1988 suggests that the future of the commercial fishery is financially secure; however, it is important to note that 36% of that year's revenue was from a compensation fund and a freight subsidy program. High whitefish prices in 1988 were also important; they were the major reason for the 32% increase in real income since 1980.

BENTHIC INVERTEBRATES

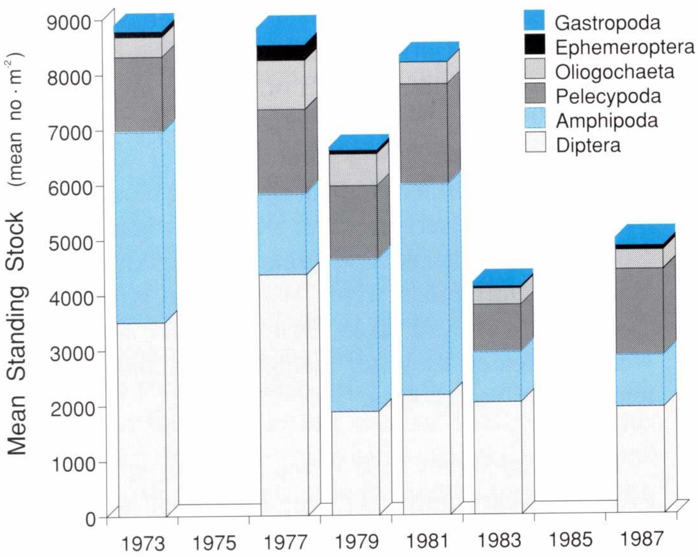
Benthic invertebrates are a major food source for bottom feeding fish. In 1973, a survey was initiated of the lakes along the diversion route and in the lower Churchill River to assess the potential effects on benthic invertebrates caused by the hydroelectric manipulations. The lakes surveyed along the lower Churchill River were: Partridge Breast, Northern Indian, and Fidler; those flooded by the Notigi Reservoir were: Karsakuwigamak, Rat, Notigi, West Mynarski, and Central Mynarski; and those below the Notigi dam were: Wapisu and Wuskwatim. Two other surveyed lakes, East Mynarski and Wood, were unaffected by water manipulations and were intended as reference sites.

The initial survey (1973) was repeated in 1977 and every two years until 1987, with the exception of 1985; the 1987 survey was conducted as part of the FEMP. Samples were taken at 3-10 sites in the profundal area of each lake and consisted of one Ekman grab per site, screened through 400μm mesh netting. Physical and chemical data were also collected. Fidler Lake was deleted in 1979 because of drying; similarly, other sites were deleted when it became hazardous to land a float plane. To examine statistical differences in invertebrate populations before and after diversion impacts, analyses of variance were conducted on lakes, or groups of lakes, with sufficient stations.

The lakes surveyed along the lower Churchill River had reduced flow, and greatly reduced volume and surface area, but maintained a similar water exchange rate. These lakes suffered a decline of one-half of the profundal invertebrate standing stock over a decade, in addition to losing approximately 46% of their area (almost the entire productive littoral zone) (Fig. 2.34). Although a concentrating effect may have occurred initially from the littoral to the profundal zone, with the invertebrates moving into deeper water like the fish of Cross Lake, the majority of benthic invertebrates were limited by available habitat, their size and swimming ability, and likely desiccated in the dried littoral zone. Establishment of a new littoral zone has been hampered by fluctuations of water released from the Missi Falls dam, and therefore it is unlikely that these lakes will regain their productivity. Like the fish of Cross Lake, the benthic invertebrates of the lower Churchill River will maintain reduced populations, primarily below the range of water level fluctuations.

Trends in total standing stocks varied with each lake as

Figure 2.34 *Mean Standing Stock of Benthic Invertebrates in the Lower Churchill River Lakes, 1973 - 1987*



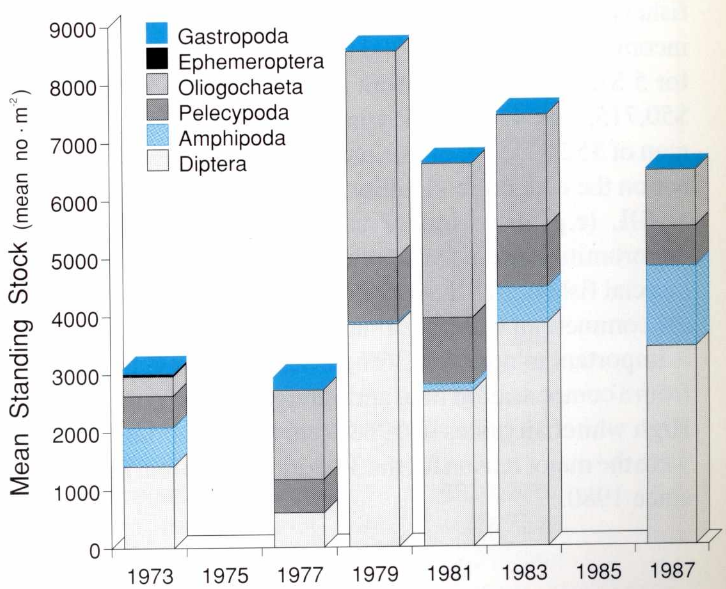
follows: 1) Partridge Breast Lake - decreased from 23743 organisms m^{-2} in 1973 to 6740 organisms m^{-2} in 1987; 2) Northern Indian Lake - 5585 organisms m^{-2} in 1973, followed by a temporary increase in the late 1970s to 35-48% above pre-impact levels, falling to approximately one-half of pre-impact values in 1983; 1987 values were lower than those of 1973; 3) Fidler Lake - standing stocks in the initial years resembled those of Northern Indian Lake, however continued sampling was impossible because of desiccation of much of Fidler Lake. Two immediate responses of invertebrate taxa to lowered water flow involved an immediate decline in number of Amphipoda and a temporary increase in Ephemeroptera numbers. For most taxa, the overall trend one decade after the impact occurred was one of decline to abundances lower than before flow manipulations.

Notigi Reservoir followed a sequence of events which is typical for a new reservoir. Effects ranged from deoxygenation of the hypolimnion, to increases in dissolved nutrients and particulate organic matter derived from flooded vegetation, to a resulting bloom of phytoplankton, bacteria and invertebrates; these lower trophic levels support higher trophic levels. Productivity brought on initially by the additional nutrients and particulate food resulted in increases in benthic invertebrates to levels seen only in the lower Churchill River before damming. One major difference in the response of Notigi Reservoir in comparison to most new reservoirs is that it has not, yet, experienced a subsequent decline in populations of benthic invertebrates. Although total benthic standing stocks declined slightly in later years, the anticipated decrease had not occurred by 1987, likely because the Churchill River itself was a major source of nutrients to the reservoir. (Fig. 2.35). Benthic invertebrate productivity is likely to remain high in this reservoir particularly with the continued input of nutrients from the diverted Churchill River, the relatively small drawdown, and the eventual development of stable benthic habitat as shoreline erosion ends and flooded vegetation disappears.

The faunal composition in 1973 of the lakes that formed the Notigi Reservoir was diverse, and considerable numbers of the six major taxa were present. The composition seen in 1977, however, was substantially different: the Amphipoda and the Ephemeroptera disappeared from all of the lakes after initial flooding, leaving the Diptera, Pelecypoda, and Oligochaeta to provide most of the benthic abundance. By 1981, the standing stock was down approximately 25% but species diversity was returning, and Amphipoda were again regularly found in the samples. By 1987, taxa richness overall reflected the pre-impoundment conditions of 1973, with all of the taxa except the Pelecypoda having almost doubled in abundance.

Wapisu and Wuskwatim lakes experienced a similar sequence of events in which an initial high, post-impact

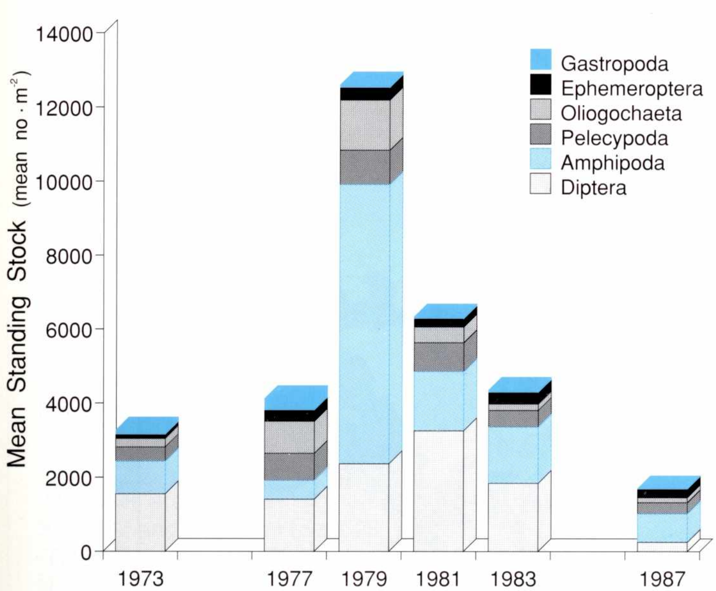
Figure 2.35 *Mean Standing Stock of Benthic Invertebrates in the Notigi Reservoir Lakes, 1973 - 1987*



productivity, based largely on benthic organisms feeding on drowned vegetation, was followed by a trophic depression when organic matter was exhausted or rendered unavailable by siltation. (Fig. 2.36). However, the extent of the changes differed between these two lakes. Wapisu Lake nearly always had higher standing stocks than Wuskwatim Lake; pre-impoundment totals were 2.5 times higher. With the influx of Churchill River water in 1977, and consequent increase in depth, the diverse benthic fauna responded by increasing in total abundance to 12,600 organisms m^{-2} in 1979 from 4245 m^{-2} in 1973. By 1987 total standing stocks had decreased to 1602 organism m^{-2}. The proportions of taxa in 1987 were similar to those of 1973, but with fewer Diptera.

Total standing stocks in Wuskwatim Lake doubled in 1977 to 3197 organism m^{-2} from 1644 m^{-2} in 1973; they subsequently declined to 1819 m^{-2} in 1987. Although the fauna of Wuskwatim Lake was less abundant than in Wapisu Lake, the same taxa were present. Proportionally, however, Ephemeroptera and Gastropoda were more abundant in this benthic community. The response of the benthic community in this lake was similar to that in Wapisu Lake but of lower amplitude. The intervening Threepoint Lake and approximately 55 km of river between the two lakes may have moderated the response of the invertebrates in Wuskwatim Lake.

Figure 2.36 Mean Standing Stock of Benthic Invertebrates in Wapisu and Wuskwatim Lakes, 1973 - 1987



The two reference lakes, Wood and East Mynarski, showed some variation in standing stocks over time, but most especially East Mynarski Lake. Total numbers in East Mynarski Lake fluctuated considerably without apparent pattern during the course of the study; however, the total standing stock was always higher in surveys done after 1973. In Wood Lake, the total standing stocks after 1973 were not significantly different from one another; 1973 stands alone with low numbers. Total invertebrate standing stocks of this lake were usually about one-half of the riverine lakes, indicating the extent of the influence of river-borne nutrients and their effects on benthic populations.

In summary, standing stock for the lower Churchill River lakes is about 50% of pre-development levels, on a per-area basis. Loss of almost half of the most productive lake areas has meant that the present total population of benthic invertebrates is only about 25% of that before diversion, a major loss of food resources for bottom feeding fish. Standing stock in the Notigi Reservoir has stabilized at about twice the pre-impact levels, which combined with an areal increase of approximately four times, means that the present water body has approximately eight times more benthic invertebrate standing stock than previously. Future levels of standing stock for this reservoir depend on the continued supply of nutrients from the Churchill River, but should remain higher than pre-impoundment levels.

WATERFOWL

Concerns expressed by the Nelson House Band members that opportunities to hunt waterfowl in the fall had declined after the diversion of the Churchill River resulted in a FEMP study of waterfowl in this area. The purpose of this study was to investigate current waterfowl use in the fall and to the extent possible, compare it with historical usage.

One-on-one interviews were conducted with 7 waterfowl hunters in Nelson House on February 19, 1988. Hunters were asked to describe their waterfowl hunting experiences and to share their observations on changes in the numbers and distribution of waterfowl; this information was used in the determination of the timing and location of the waterfowl surveys in this area. On September 1, 9, and 26, 1988, a series of low-altitude aerial surveys were flown using a Bell 206B helicopter with one observer. Survey routes followed the shorelines of

Wapisu, Threepoint, Wuskwatim and Honeymoon lakes and the Rat-Burntwood rivers (Fig. 2.37). Notakikwaywin Lake, which is located off of the CRD was also surveyed on September 26, 1988.

A summary of the 1988 survey data are shown in Fig. 2.38. Maximum numbers and densities are recorded because they were more likely to represent the potential for waterfowl use. Low numbers of ducks and Canada geese were observed in all surveyed areas, with the exception of the single survey of Notakikwaywin Lake where duck densities were considerably higher. There was considerable variability in the numbers of ducks and Canada geese observed between surveys and sites, (e.g. along the Rat River a total of 4 ducks were observed on September 1, 1988 and 44 on September 9, 1988).

The only historical data on waterfowl use of this area are the results of a 1973 aerial survey (LWCN 1975); these data are summarized in Fig. 2.39. Honeymoon and Notakikwaywin lakes and the reach of the Burntwood River upstream of Threepoint Lake were not surveyed in 1973; other areas surveyed in 1973 are, in general, comparable with those surveyed in 1988. As was the case with the 1988 survey, there was considerable variability in the numbers of ducks and Canada geese observed between surveys and sites in 1973 (e.g. 72 dabbling ducks were observed at Wuskwatim Lake on September 7, 1973 and 944 on September 26, 1973).

Figure 2.37 Waterfowl Survey Routes in the Churchill River Diversion Area, 1988

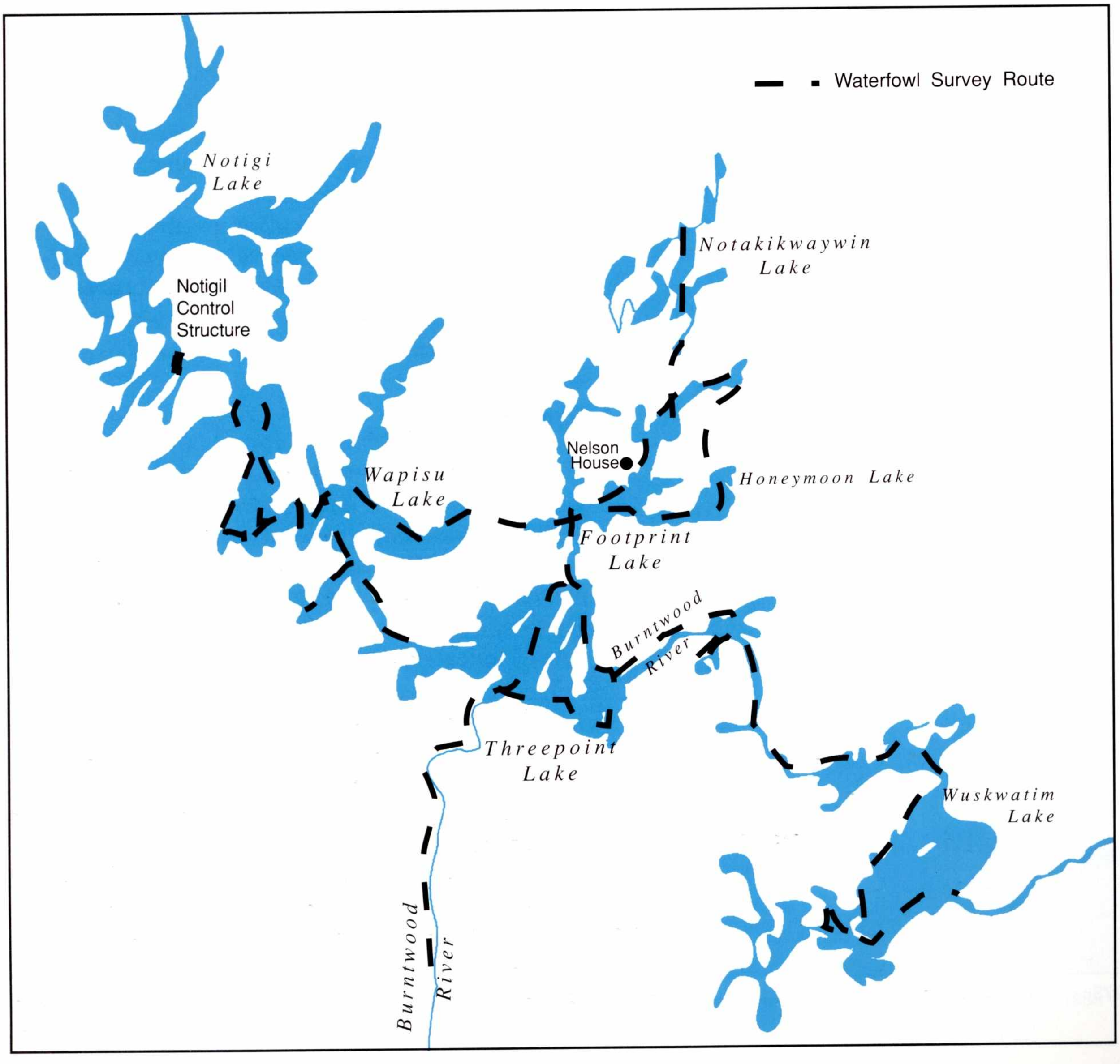


Figure 2.38 Numbers (N) and Densities[1] (D) of Waterfowl Observed in 1988 in the Churchill River Diversion Area

	Sept 1, 1988		Sept 9, 1988		Sept 26, 1988	
Dabbling Ducks	N	D	N	D	N	D
Wapisu Lake	0	0.00	25	0.65	10	0.17
Rat River	4	0.20	30	1.50	0	0.00
Burntwood River (upstream)	5	0.28	4	0.23	6	0.34
Threepoint Lake	0	0.00	0	0.00	0	0.00
Burntwood River (downstream)	1	0.03	25	0.78	0	0.00
Wuskwatim Lake	0	0.00	0	0.00	1	0.02
Honeymoon Lake	2	0.30	22	2.51	3	0.34
Notakikwaywin Lake	N.S.	N.S.	N.S.	N.S.	35	2.47
Diving Ducks	N	D	N	D	N	D
Wapisu Lake	1	0.02	2	0.05	2	0.03
Rat River	0	0.00	14	0.70	0	0.00
Burntwood River (upstream)	0	0.00	12	0.68	0	0.00
Threepoint Lake	0	0.00	0	0.00	1	0.06
Burntwood River (downstream)	0	0.00	1	0.03	0	0.00
Wuskwatim Lake	20	0.58	5	0.12	5	0.12
Honeymoon Lake	7	1.04	1	0.11	2	0.23
Notakikwaywin Lake	N.S.	N.S.	N.S.	N.S.	27	1.91
Total Ducks[2]	N	D	N	D	N	D
Wapisu Lake	1	0.02	27	0.70	12	0.20
Rat River	4	0.20	44	2.20	0	0.00
Burntwood River (upstream)	5	0.28	16	0.91	6	0.34
Threepoint Lake	5	0.55	0	0.00	1	0.06
Burntwood River (downstream)	1	0.03	26	0.81	0	0.00
Wuskwatim Lake	20	0.58	5	0.12	6	0.14
Honeymoon Lake	9	1.33	23	2.63	5	0.57
Notakikwaywin Lake	N.S.	N.S.	N.S.	N.S.	62	4.38
Canada Geese	N	D	N	D	N	D
Wapisu Lake	0	0.00	0	0.00	0	0.00
Rat River	8	0.40	2	0.10	0	0.00
Burntwood River (upstream)	0	0.00	0	0.00	0	0.00
Threepoint Lake	0	0.00	0	0.00	0	0.00
Burntwood River (downstrem)	0	0.00	0	0.00	0	0.00
Wuskwatim Lake	0	0.00	75	1.74	0	0.00
Honeymoon Lake	0	0.00	0	0.00	0	0.00
Notakikwaywin Lake	N.S.	N.S.	N.S.	N.S.	0	0.00

[1]Densities are birds per kilometer based on survey lengths as follows: 181.6 km (Sept 1, 1986); 169.1 km (Sept 9, 1986); and 196.4 km (Sept 26, 1986).

[2]Total ducks includes diving ducks, dabbling ducks, and unidentified ducks.

N.S. = not surveyed

X = highest density by area

Figure 2.39 Numbers of Waterfowl Observed in 1973 in the Churchill River Diversion Area

Dabbling Ducks	Sept 7, 1973	Sept 26, 1973
Wapisu Lake	414	46
Rat River	200	87
Threepoint Lake	15	377
Burntwood River (downstream)	57	502
Wuskwatim Lake	72	944
Diving Ducks		
Wapisu Lake	12	22
Rat River	10	0
Threepoint Lake	20	31
Burntwood River (downstream)	5	0
Wuskwatim Lake	265	169
Total Ducks[1]		
Wapisu Lake	426	95
Rat River	235	87
Threepoint Lake	35	408
Burntwood River (downstream)	97	502
Wuskwatim Lake	388	1127
Canada Geese		
Wapisu Lake	0	0
Rat River	90	20
Threepoint Lake	6	90
Burntwood River (downstream)	0	0
Wuskwatim Lake	12	0

[1]Total ducks includes diving ducks, dabbling ducks, and unidentified ducks.

N.B. Densities are not shown because length of shorelines surveyed are not available.

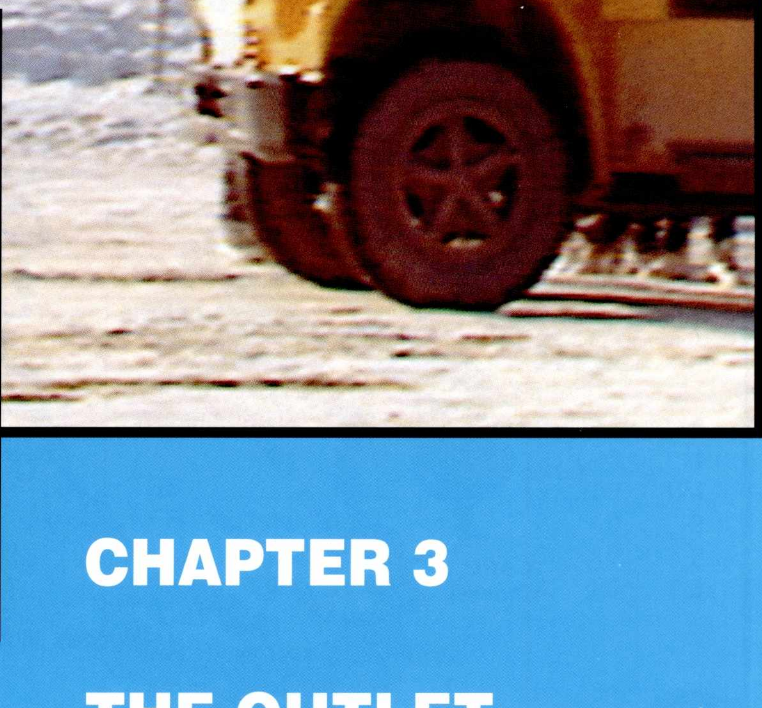
Comparison of the 1973 and 1988 data suggest that dabbling ducks declined by 97%, diving ducks declined by 89%, and Canada geese declined by 57%. It is risky, however, to draw broad conclusions about waterfowl numbers on the basis of the 1973 and 1988 surveys, because both surveys provide only a limited number of "point-in-time" counts. Furthermore, large between-year changes are not uncommon in waterfowl counts derived from aerial surveys. In addition, the small numbers of diving ducks and Canada geese observed in both surveys make it difficult to state, definitively, that a decline has occurred in all survey areas.

Two possible causes for the apparent change in dabbling duck use of the area were considered: 1) changes in regional populations; and 2) modifications to the local physical environment and waterfowl habitat. The only data on regional populations are the annual breeding pair surveys conducted by the U.S. Fish and Wildlife Service (USF & WS) in northern Manitoba. These data provide adjusted population estimates for ducks and serve as general indicators of population size and trends. These data estimate the size of breeding populations; they do not indicate the size of fall migrating flocks.

Data from Stratum 24, the USF & WS region, which most closely approximate the FEMP study area, were reviewed for the periods 1964 - 1975 and 1977 - 1988. When the 12-year means for the two periods were compared, the numbers of dabbling ducks were marginally higher (0.2%) in the latter period. However, large year-to-year fluctuations occurred during these periods; thus, the means on which these changes were based would vary somewhat as individual year counts were added or subtracted. The apparent reduction in the dabbling duck use of the diversion area does not appear to be a reflection of a long-term decline in regional populations as represented by the Stratum 24 counts, since no such trends were evident in the stratum data.

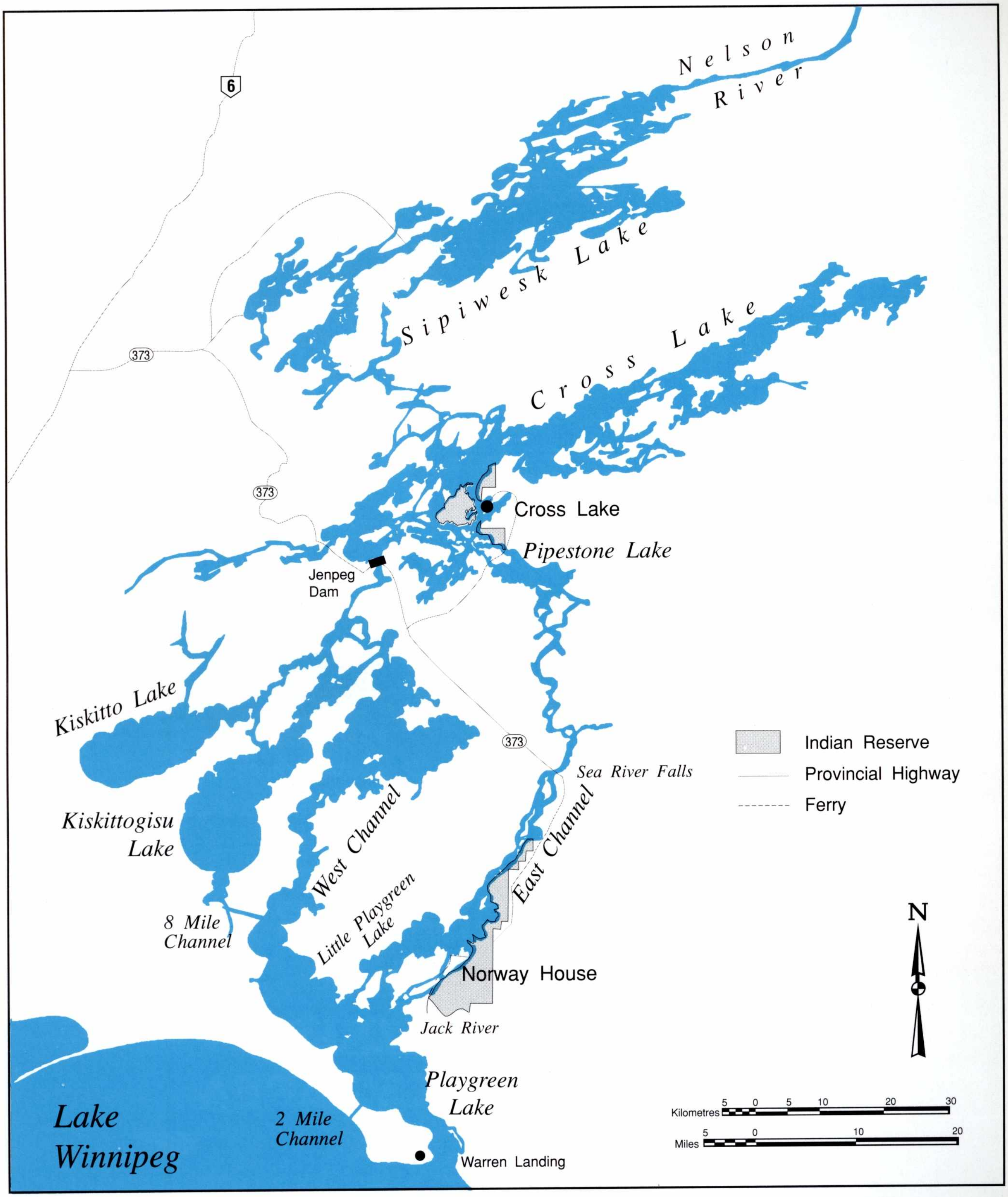
Modifications to the local physical environment and waterfowl habitat was the second possible cause considered for the apparent decrease in dabbling ducks. As noted previously in this chapter, the extensive flooding along the diversion route greatly altered the shoreline characteristics (e.g. increased debris and standing dead trees, etc.); this alteration has created poor quality habitat for waterfowl, rendering these areas less attractive for dabbling ducks and (to a lesser extent) for diving ducks.

In summary, low numbers of ducks and Canada geese were observed in September 1988 in the Churchill River diversion area. Temporal and spatial data, as well as the current data, were too limited to estimate changes in the levels of waterfowl use of this area with any degree of confidence. The data suggest, however, that there may have been a decline in the dabbling duck use of this area following diversion.




CHAPTER 3

THE OUTLET LAKES AREA

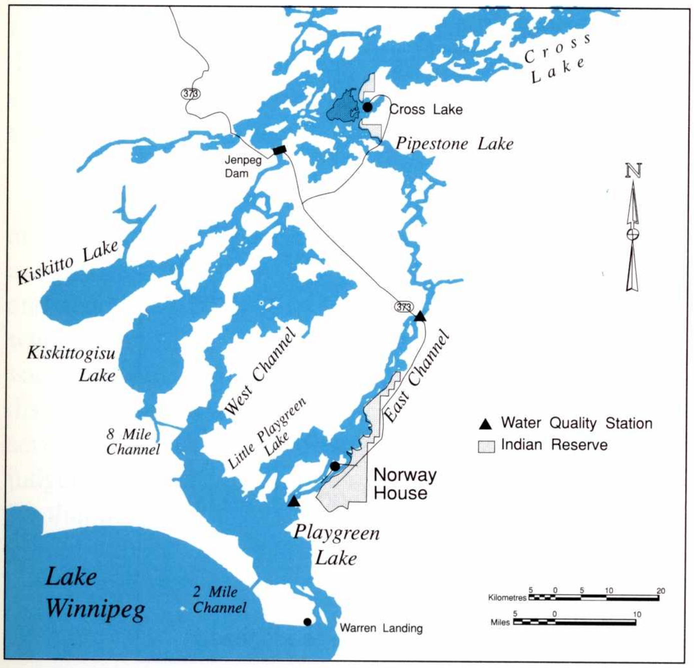

Nelson River
6
Sipiwesk Lake
Cross Lake
373
373
Cross Lake
Pipestone Lake
Jenpeg Dam
Kiskitto Lake
373
Indian Reserve
Provincial Highway
Ferry
Sea River Falls
Kiskittogisu Lake
West Channel
East Channel
8 Mile Channel
Little Playgreen Lake
N
Norway House
Jack River
Playgreen Lake
Lake Winnipeg
2 Mile Channel
Warren Landing
Kilometres 5 0 5 10 20 30
Miles 5 0 10 20

MORPHOLOGY AND SEDIMENT REGIMES

FEMP investigations of the morphology and sediment regimes in the Outlet Lakes area were limited to: 1) a review of the existing literature on the morphology of this area, most of which was the product of work conducted by or for Manitoba Hydro; 2) an assessment of future required morphological study; and 3) the collection of data on suspended sediment, as part of the FEMP water quality program, at two sites, the Nelson River below Sea River Falls and Jack River (Fig. 3.1). (Cross Lake was not sampled under FEMP because it was one of the six lakes along the Nelson, Rat, and Burntwood rivers that were being extensively monitored by the Manitoba Department of Natural Resources, concurrent with FEMP.)

Regular monitoring of the 8-Mile and 2-Mile channels, located in relatively low-lying, flat areas of peat and clay, indicated apparently substantial erosion during the first two years of operation, but much less since. Near the entrance to 2-Mile Channel, the protective beach found elsewhere on the north shore of Lake Winnipeg is missing and rates of bank retreat are much higher. Since the erosion process is closely related to the occurrence of severe storms, it is episodic. In Playgreen Lake, comparative analysis of air photos, dating back to 1946, for 8 sites showed highly variable bank retreat rates. No changes in shore erosion rates attributable to Lake Winnipeg regulation could be detected.

Figure 3.1 FEMP Water Quality Sites in the Outlet Lakes Area



Future morphological and related studies that should be considered for the Outlet Lakes area include: 1) documentation of the large erosion rates observed at the entrance to the 2-Mile Channel and an assessment of their long-term effects; 2) an investigation of the sedimentation in Playgreen Lake off the mouth of 2-Mile Channel; and 3) an examination of the morphological evolution of the lakes in this area, especially Cross and Sipiwesk lakes.

Data on turbidity, total dissolved solids (TDS), non-filterable residue (NFR) and colour for the two sites in the Outlet Lakes area are shown in Fig. 3.2. As was the case for all the sites sampled under the FEMP water quality project, the mean turbidity values for these 2 sites exceeded the Canadian Water Quality Guideline for turbidity levels in drinking water of 5 NTU. Statistical comparison of current conditions with pre-development conditions was not possible for either site because of the lack

Figure 3.2 Turbidity, Total Dissolved Solids, Non-Filterable Residue, and Colour for the Outlet Lakes Area, 1987 - 1989

Variable	Type	Units	Mean	SD	Max.	Min.
Nelson River below Sea River Falls						
Turbidity	field	NTU	10.+	3.	16.+	6.+
	lab	NTU	8.+	3.	13.+	4.
TDS		$mg \cdot L^{-1}$	163.	19.	189.	124.
NFR	total	$mg \cdot L^{-1}$	6.	3.	10.	3.
	fixed	$mg \cdot L^{-1}$	5.	3.	10.	2.
Colour	true	relative	9.	4.	20.+	3.
Jack River (West Channel)						
Turbidity	field	NTU	10.+	6.	34.+	5.
	lab	NTU	9.+	5.	26.+	3.
TDS		$mg \cdot L^{-1}$	165.	16.	185.	122.
NFR	total	$mg \cdot L^{-1}$	7.	4.	20.	2.
	fixed	$mg \cdot L^{-1}$	5.	4.	17.	1.
Colour	true	relative	8.	4.	15.	3.

Footnotes
\+ Exceeds limit for drinking water
Exceedences in the "Max." column indicates occasional exceedences.

of sufficient pre-development data. However, an analysis by the Manitoba Department of Environment of the more extensive database for their nearby site on the Nelson River near Norway House showed a small increase in turbidity from the 1972 - 1976 period to the 1976 - 1984 period.

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; 2 of these 11 sites were located in the Outlet Lakes area. This section presents the results of the water quality study for this 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 exists; 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 the NFC communities.

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.

Nelson River Below Sea River Falls

A few variables at this site (Figs. 3.2 + 3.3) exceeded the Canadian Water Quality Guidelines; these same variables commonly exceeded the Guidelines at most of the FEMP water quality sites, including those on the unregulated rivers. Mean levels of turbidity, extractable aluminum (Al) and total copper (Cu) exceeded the Guidelines. While extractable iron (Fe) concentrations

Figure 3.3 Water Quality Variables Which Exceeded the Canadian Guidelines in the Outlet Lakes Area, 1987 - 1989

Variable	Type	Units	Mean	SD	Max.	Min.
Nelson River below Sea River Falls						
Al	extractable	$mg \cdot L^{-1}$	0.181^{++}	0.094	0.385^{++}	<0.100
Cu	total	$mg \cdot L^{-1}$	0.0047^{++}	0.0054	0.0208^{++}	0.0011
Fe	extractable	$mg \cdot L^{-1}$	0.197	0.072	0.365$^{+,++}$	0.087
Mn	extractable	$mg \cdot L^{-1}$	0.016	0.012	0.067^{+}	0.005
Jack River (West Channel)						
Al	extractable	$mg \cdot L^{-1}$	0.176^{++}	0.126	0.442^{++}	<0.100
Cu	total	$mg \cdot L^{-1}$	0.0034^{++}	0.0038	0.0159^{++}	0.0011
Fe	extractable	$mg \cdot L^{-1}$	0.187	0.106	0.457$^{+,++}$	0.070
Pb	total	$mg \cdot L^{-1}$	0.0007	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.

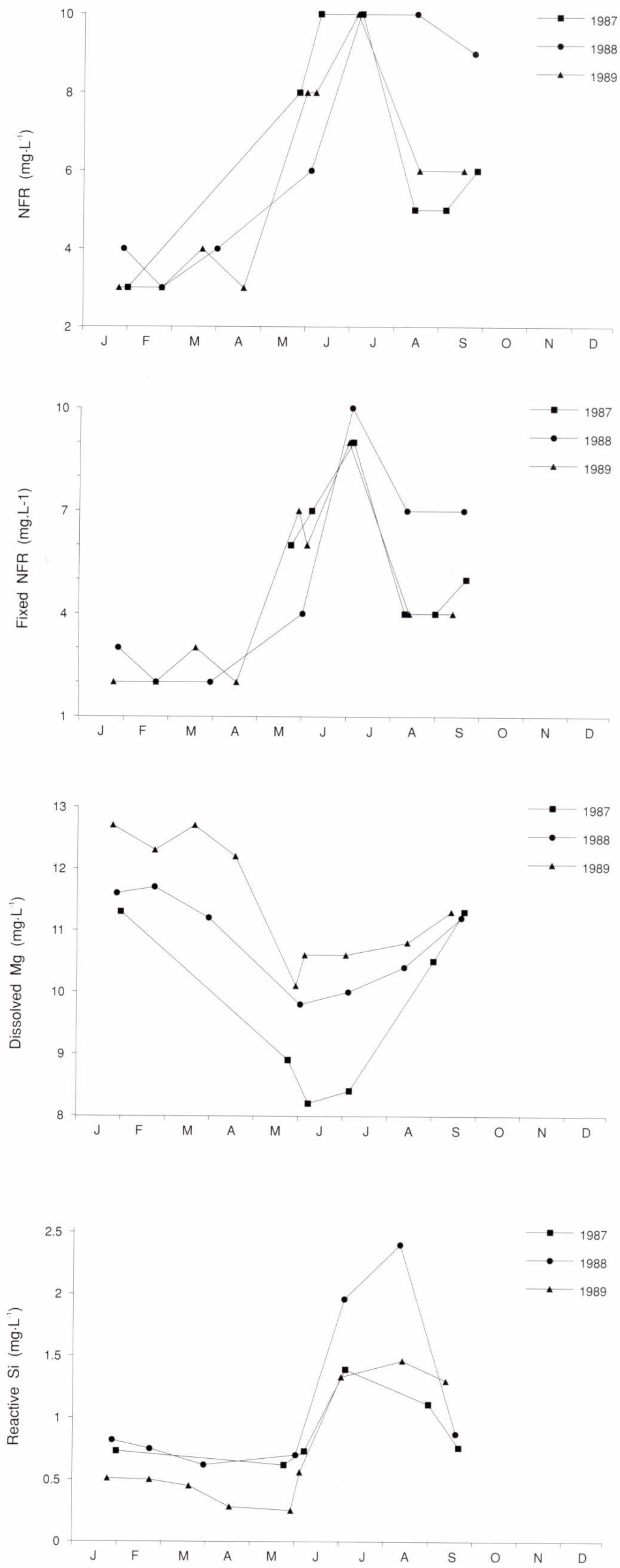
occasionally exceeded the Guidelines, they were lower at this site than for those along the Churchill River Diversion and in the natural rivers. Fe concentrations above the GUidelines were characteristic of natural rivers in northern Manitoba, occurring at all of the sites samples in this study. Extractable manganese (Mn) concentrations at this site also occasionally exceeded the Guidelines.

Insufficient pre-development data for this site prevented a quantitative analysis of water quality changes from pre-development to post-development conditions. The only way of assessing the impacts of the LWCN project on the water quality at the Sea River Falls site would be to use data from an acceptable surrogate site. The Nelson River at Norway House is the only possible choice for this purpose as it was routinely sampled from 1972 to 1974; however, it would first be necessary to demonstrate that water quality at the two sites should have been the same prior to development. This validation would require some field reconnaissance as well as consultation with individuals knowledgable in the recent history of development (i.e. past 25 years) along this reach of the Nelson River.

A number of water quality variables were correlated with Nelson River discharge at this site during the 1987 - 1989 study period. Negative correlates included most major ions, alkalinity and conductivity, while positive correlates included turbidity, non-filterable residue (NFR) and extractable Fe. Most of the significant water quality - discharge relationships at this site were strong enough to offer promise in impact prediction and management. The sparse pre-development water quality data precluded an analysis of changes in discharge relationships since the LWCN project.

Seasonal differences, for the 1987 - 1989 study period at this site, were limited, with only 4 variables showing any seasonal variation (Fig. 3.4). Total and fixed NFR and reactive silicon (Si) were lower under ice-cover while magnesium (Mg) was higher. While 3 of these variables were correlated with discharge, fluctuations in discharge fail to account for the seasonal patterns observed at this site. The pre-development data set was judged inadequate for an assessment of seasonal patterns for that period.

Figure 3.4 Seasonal Water Quality Trends for Selected Water Quality Variables in the Nelson River below Sea River Falls



Jack River

The water quality variables at the Jack River site that exceeded the Canadian Water Quality Guidelines were similar to those at the Sea River Falls site. Mean levels of turbidity, extractable Al, and total Cu exceeded the Guidelines, while levels of extractable Fe and total lead (Pb) occasionally exceeded these Guidelines. (Figs. 3.2 + 3.3)

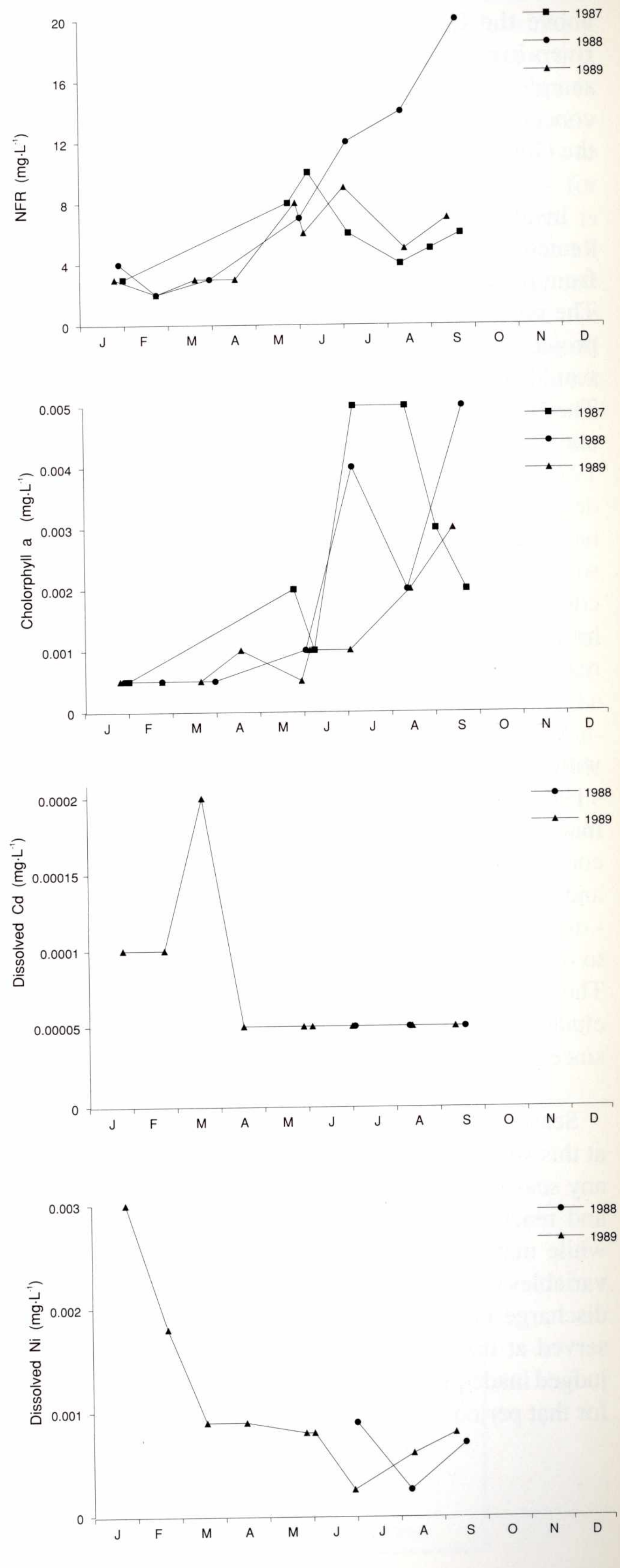
There are no pre-development water quality data for this site and consequently no means of directly determining to what extent, if any, the water quality has changed since the LWCN project. As suggested for the Sea River Falls site, it may be possible to use pre-development data for another upstream site as a surrogate. The only upstream site which was adequately sampled prior to development is the Nelson River at Warren Landing, which has been sampled since 1969. Verification of this site as a suitable surrogate would be required.

There were few clear differences in water quality between the ice-free and ice-cover seasons during the 1987 - 1989 FEMP study. NFR was lower under ice-cover, as was chlorophyll a, while dissolved cadmium (Cd) and nickel (Ni) concentrations were higher under ice-cover (Fig. 3.5). The lower chlorophyll a under ice was consistent with the trend at other sites. The lower NFR and higher Cd and Ni under ice could be a product of lower winter flows, but this cannot be confirmed due to the lack of discharge data for this site.

Bacteriological Assessment

A bacteriological assessment of the recreational waters in the Norway House and Cross Lake areas was conducted in 1987. Seasonal studies were conducted from May to September at 8 sites at Norway House and 5 sites at Cross Lake; in addition, intensive studies were conducted in the vicinity of the water treatment plants in August. The geometric mean fecal coliform bacteria for the seasonal studies ranged between 3 and 39 per 100 mL at Norway House and 6 and 44 at Cross Lake. The geometric means for the intensive study were 9 per 100 mL for both Cross Lake and Norway House. Comparison of the results of the intensive studies to the Canadian Water Quality Guidelines of 200 bacteria per 100 mL indicates that the bacteriological water quality at these sites were generally acceptable for recreational water

Figure 3.5 Seasonal Water Quality Trends for Selected Water Quality Variables in the Jack River near Norway House



uses. Limited sampling frequency for the seasonal studies precludes direct comparisons of their results to this guideline.

MERCURY

The FEMP mercury program focused on the Churchill River Diversion (CRD) route since this was the region within the FEMP study area that has experienced the most severe mercury problem. The basic processes that govern the production of methyl mercury, the expected longevity of elevated mercury levels in fish, and possible mitigative measures were discussed in chapter 2. In this chapter, only mercury issues specific to the Outlet Lakes area are discussed, namely: 1) current mercury levels in fish in Cross and Sipiwesk lakes; 2) current mercury levels in residents of Cross Lake and Norway House; and 3) the expected future mercury regime for Cross and Sipiwesk lakes.

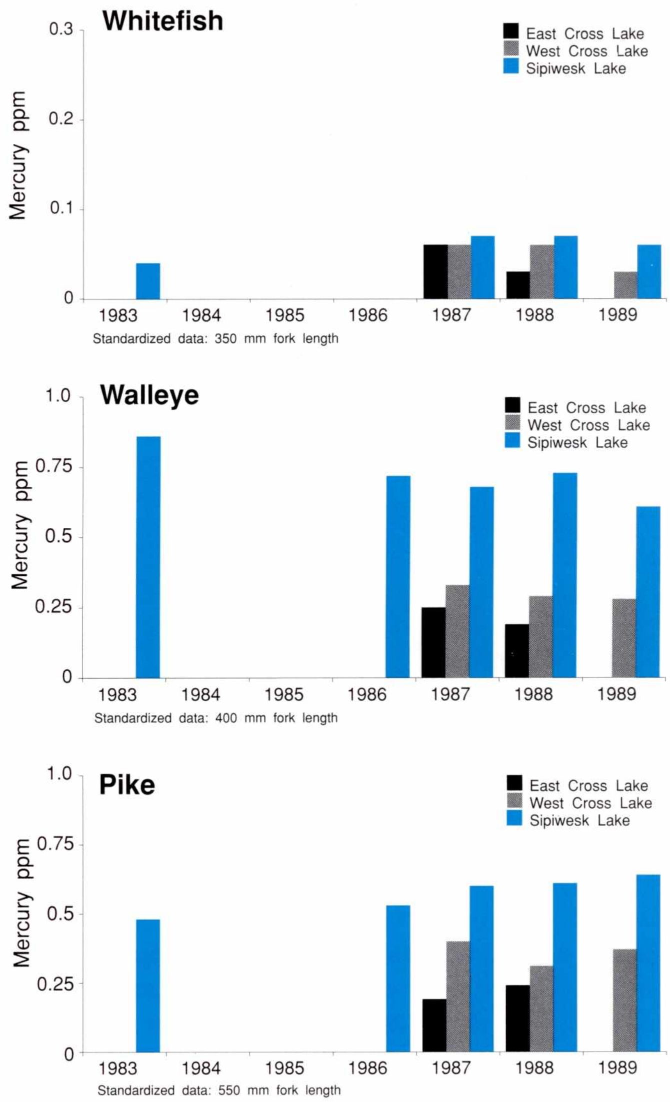
Mercury levels in fish in Cross and Sipiwesk lakes were sampled by the Manitoba Department of Natural Resources (Fig. 3.6). Mercury levels in whitefish, walleye and pike from Cross Lake were all below the Canadian guideline of 0.5 ppm. In Sipiwesk Lake, mercury levels in whitefish were also below the 0.5 ppm level, while mercury levels in walleye and pike were generally between 0.5 ppm and 1.0 ppm.

Mercury levels in residents of Cross Lake and Norway House were monitored by Health and Welfare Canada (Fig. 3.7). Of the approximately 20% of the population of Cross Lake tested for mercury in 1989/90, 98% had values in the 0-19 ppb range, which is considered to be the "normal" range; all of the remaining values were in the range of 20-50 ppb. For women of child-bearing age who were tested, 98% had values in the 0-19 ppb range.

Of the approximately 24% of the population of Norway House tested in 1988/89, 97% had values in the range of 0-19 ppb; all of the remaining values were in the 20-50 ppb range. For women of childbearing age, almost 98% were in the 0-19 ppb range. Unlike South Indian Lake and Nelson House, there were never any mercury levels in the tested populations of Cross Lake and Norway House that exceeded 99 ppb.

The residents of Cross Lake have expressed a concern that the new Cross Lake weir, which will raise lake levels during the open-water season to more closely approxi-

Figure 3.6 Standardized Mean Muscle Mercury Concentrations for Fish in Cross and Sipiwesk Lakes, 1983 - 1989



mate pre-development levels, will cause a rise in mercury levels in fish. However, unlike the CRD, there will be no extensive backflooding of bog or forest terrain associated with the weir, because the rise in water level is expected to be contained within the natural basin of Cross Lake. It is therefore unlikely that the resulting higher water level on Cross Lake will cause any long-term increase in fish mercury levels.

Another important factor in assessing the potential of the Cross Lake weir to cause increased mercury levels in fish is the fact that the majority of the shoreline of Cross Lake is bedrock controlled, which greatly limits the potential for revegetation of the drawdown zone during

Figure 3.7 Mercury Levels in Residents of Cross Lake and Norway House

Mercury Levels in Residents of Cross Lake

Year	*Sample Number*	*0 - 19 ppb*	*20 - 99 ppb*	*>99 ppb*
1977	96	79	17	0
1978	1	1	0	0
1979	0	0	0	0
1980	70	58	12	0
1981	225	207	18	0
1982	296	275	21	0
1983	7	7	0	0
1984	200	192	8	0
1985	93	90	3	0
1989-90	494	484	10	0

1989-90 Groups	*Sample Number*	*0-19 ppb*	*20-50 ppb*	*50-80 ppb*
Men	92	87	5	0
Women:				
12-45 yrs	136	133	3	0
over 45 yrs	58	56	2	0
Children:				
1-4 yrs	33	33	0	0
5-12 yrs	175	175	0	0

Mercury Levels in Residents of Norway House

Year	*Sample Number*	*0 - 19 ppb*	*20 - 99 ppb*	*>99 ppb*
1977	144	127	17	0
1978	81	75	6	0
1979	0	0	0	0
1980	66	59	7	0
1981	107	99	8	0
1982	127	126	1	0
1983	54	54	0	0
1984	221	212	9	0
1989-90	775	752	23	0

1989-90 Groups	*Sample Number*	*0-19 ppb*	*20-50 ppb*	*50-80 ppb*
Men	326	318	8	0
Women:				
12-45 yrs	275	269	6	0
over 45 yrs	94	85	9	0
Children:				
1-4 yrs	46	46	0	0
5-12 yrs	34	34	0	0

periods of low water. Cross Lake levels have fluctuated substantially since the Jenpeg Generating Station began operating in 1976, by as much as 3 m in a single year, yet mercury levels in pike and walleye from Cross Lake are not higher than the average for lakes in northern Manitoba (0.3-0.4$\mu g \cdot g^{-1}$). Any vegetation which became established in the drawdown zone apparently was not of sufficient quantity to cause a substantial elevation in net microbial methylation when it was flooded. Similar conditions would likely prevail after the weir is constructed, in which case an increase in fish mercury levels would not occur. This assessment should be confirmed, however, by mercury monitoring for at least 2 years after the weir is constructed.

The situation in Cross Lake contrasts with that of Sipiwesk Lake where only 5% of the post-impoundment shoreline is bedrock controlled, with the remainder composed of flooded forest and wetland soils which provide excellent substrate for growth of vegetation during periods of low water. In the drawdown zone of Sipiwesk Lake, recolonization by grasses and other fast growing vegetation occurs. These materials are flushed into the lake during periods of high water, becoming available to the methylating bacteria. A continuous source of fresh organic matter such as this can be expected to maintain methyl mercury production at a high level longer than would be the case otherwise. This appears to be the case in Sipiwesk Lake, where the specific methylation rate in the flooded zone was at the high end of the expected range in 1987 and where mercury levels in fish were greater than the average for northern Manitoba lakes.

PLAYGREEN LAKE / LAKE WINNIPEG WHITEFISH STOCK GENETICS

It has long been supposed that there are movements of whitefish between Playgreen Lake and the north end of Lake Winnipeg. For example, a tagging program in 1973 of a large number of whitefish at Warren Landing found that fish were recaptured in both Lake Winnipeg and in Playgreen Lake. Changes in the geographic and/or temporal distribution of flow between Lake Winnipeg and Playgreen Lake might be expected to disrupt migration patterns between the two lakes, and possibly also the genetic structure of stocks in the area. Since the construction of 2-Mile Channel (2MC) in 1975 and the regulation of Lake Winnipeg, the geographic and seasonal distribution of flows through the Playgreen Lake area have been altered. For example, a higher proportion of flow now proceeds by the west channel of the Nelson River and a significant proportion of the flow now leaves Lake Winnipeg via 2MC, especially during periods of low water levels on Lake Winnipeg.

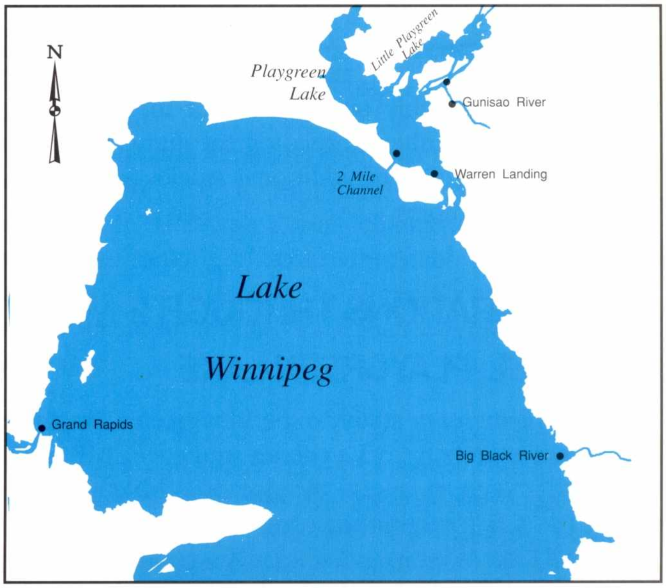
Since these changes in the water regime, fishermen on Playgreen Lake have claimed that the spatial distributions and seasonal movements of separate stocks of whitefish in the Lake Winnipeg and Playgreen Lake areas have changed. In response to these claims, a FEMP investigation was initiated of the genetic structuring of the population of whitefish in Playgreen Lake and in the northern end of Lake Winnipeg. The objective of this study was to compare the genetic stock of the whitefish populations in 1989 to that of 1975 to determine if a significant change had occurred.

A fish stock is by definition (Larkin 1972) a population which shares a common environment and gene pool. This definition implies a degree of reproductive isolation between various stocks of a particular fish species in a particular area; this reproductive isolation will generally be expected to result in genetic differences between stocks, which will tend to accumulate due to random genetic drift and/or selection. The genetic differences expected among various stocks of fish and the environmental differences expected in their habitats have allowed the development of various methods to identify and differentiate stocks; one of these methods, starch gel electrophoresis of isozymes, was used in the FEMP study.

Whitefish sampling was conducted in late October 1989, at a time close to that when whitefish spawning was expected to occur and hence a time when stocks would be expected to have segregated themselves into reproductively isolated units. Samples of approximately fifty mature whitefish were collected by gill netting from six different sites in northern Lake Winnipeg and Playgreen Lake (Fig. 3.8); four of these sites had also been sampled in 1975.

The major finding of the FEMP study was that there were at least two different stocks of whitefish in the northern Lake Winnipeg/ Playgreen Lake area. One stock utilized sites in the north basin of Lake Winnipeg for spawning (Grand Rapids and Big Black River) and a second stock utilized Little Playgreen Lake for spawning. The genetic differences observed in this study were likely attributable to both an innate homing of whitefish

Figure 3.8 Gill Net Sites for Playgreen Lake/Lake Winnipeg Stock Genetics Study



to natal spawning grounds and a degree of geographic isolation of stocks by distance. These genetic similarities and differences were also found in the 1975 study. It, thus, appears that these genetic differences were stable over time and that the genetic structure of the stocks has not changed since the regulation of Lake Winnipeg. This conclusion, however, is tentative and preliminary for a number of reasons.

While genetic differences indicate distinct stocks, genetic similarities do not prove identity. Thus, there may be more than two genetic stocks present in the northern Lake Winnipeg/Playgreen Lake area. This uncertainty is in part due to the need for more refined genetic models which would enable the interpretation of a more extensive data set than that which was analyzed in the FEMP study.

This uncertainty may be resolved by a planned analyses of morphological data on both the 1975 and 1989 samples. Morphological differences can exist between stocks which are not based solely on genetic differences, but are at least in part environmentally induced; this is especially true of some fish, such as whitefish, in which extreme morphological variations are possible. (This characteristic of whitefish formed the basis for a study of whitefish below Missi Falls, discussed in chapter 2.)

The morphological results may allow for the identification of additional stocks not ascertained by genetic analysis. In fact, it is already known that there were more than two genetic stocks of lake whitefish in the northern Lake Winnipeg/Playgreen Lake area in 1975; Kristofferson and Clayton (1990) found that the Grand Rapids, Big Black River, and Little Playgreen spawning stocks were separable on the basis of morphological characteristics. Further morphological study may result in the discrimination of additional stocks in the post-development samples.

FISH MIGRATIONS THROUGH 2-MILE CHANNEL, PLAYGREEN LAKE

In a complementary study to the Playgreen Lake/Lake Winnipeg whitefish stock genetics study, a FEMP study was undertaken to assess the use of 2-Mile Channel (2MC) as a fish migration corridor between Playgreen Lake and Lake Winnipeg. In the early summer (late June) and fall (mid-October) of 1989, a dual-beam echosounder was employed in fixed and mobile configurations in the 2MC area of Playgreen Lake to evaluate its utilization by migrating fish. Gill nets were set in the area of the 2MC outlet during the early summer sampling. Due to problems with software for analysis of the echosounder digitized data, echosounder information has not yet been statistically analyzed. A final project report should be available in 1992.

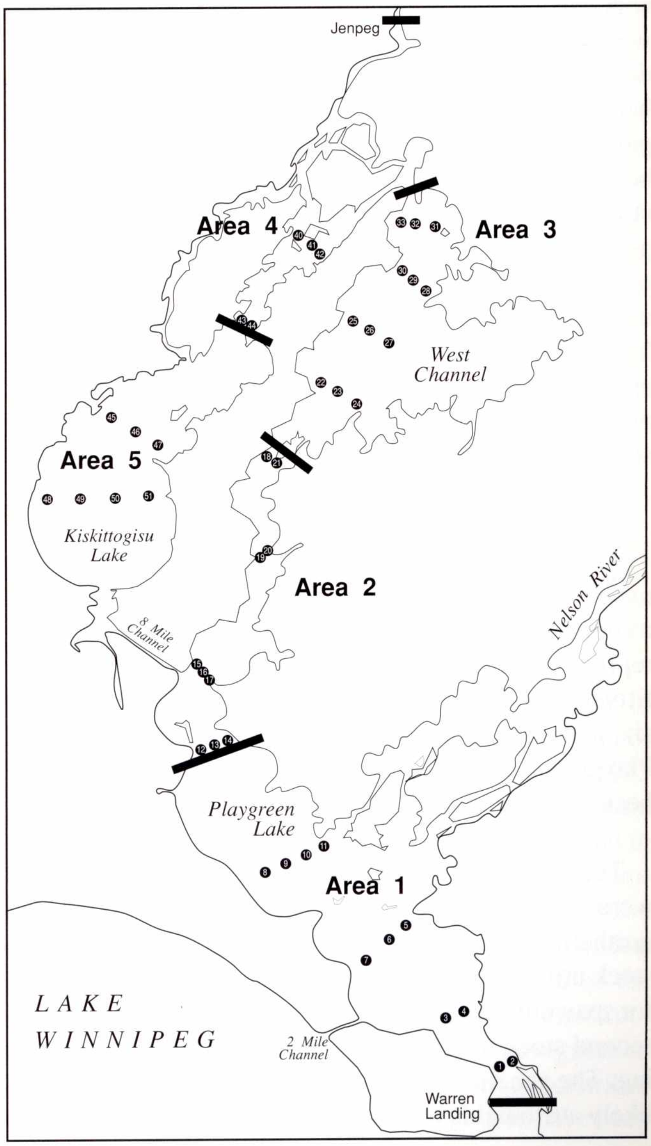
BENTHIC INVERTEBRATES

The water regime of Playgreen and Kiskittogisu lakes changed after the regulation of Lake Winnipeg. In particular, 2-Mile Channel (2MC) now contributes most of the flow to Playgreen Lake during periods of low water levels; it also introduces material that has eroded from the north shore of Lake Winnipeg into the south basin of Playgreen Lake. 8-Mile Channel (8MC), connecting the southern portions of Playgreen and Kiskittogisu lakes, enables Kiskittogisu Lake to contribute approximately 30% of the flow through the West Channel. A FEMP study was conducted to examine the status of the benthic invertebrates in Playgreen and Kiskittogisu lakes following these changes in the water regime.

Thirty-three stations were sampled in Playgreen Lake, in July 1987 and 1989, with an approximately even distribution among three areas of the lake (Fig. 3.9). Twelve stations were sampled in Kiskittogisu Lake, in July 1989, with an approximately even distribution between its two areas. Sites were located in critical areas of the lakes (e.g. just north of 8-Mile Channel) and at locations close to sites that were sampled in a 1971 benthic invertebrate survey. Three samples were taken at each site, accompanied by a Secchi disc reading, a temperature profile, and a qualitative description of the sediments, including particle size and sediment compactness. Water samples were also taken from selected stations for routine chemical analysis during the 1987 Playgreen Lake survey.

The mean standing stock in the western section of the south basin of Playgreen Lake, the area located in the sediment plume from 2MC, was more productive in 1987 than either of the central or eastern sections (13100 versus approximately 10600 organisms m^{-2}). A similar

Figure 3.9 Benthic Invertebrate Sampling Sites in the Outlet Lakes Area, 1987 and 1989



but less pronounced relationship was observed in 1989. The region north of 8MC showed no significant differences between west, central and east sections in either of the years of survey. The mean standing stock for the north basin of Playgreen Lake was less than in the south basin, especially in 1987.

While a north/south basin difference in mean standing stocks had existed in the pre-project survey conducted in 1971, this difference was intensified in 1987. In addition, the mean standing stocks were much lower for all parts of Playgreen Lake in 1971 than in either 1987 or 1989 (Fig. 3.10); this difference, however, might have been caused by methodological differences in the surveys. The lower numbers in 1989 in comparison to 1987 were probably the result of drought conditions. (Fig. 3.11)

Figure 3.10 Mean Standing Stock of Invertebrates in Playgreen and Kiskittogisu Lakes, 1971, 1987, and 1989

	Playgreen Areas			Kiskittogisu Areas	
Survey Years	**1**	**2**	**3**	**4**	**5**
1971	502	437	419	212	209
1987	11558 (1548)	8667 (1687)	4941 (419)	NS	NS
1989	4417 (1161)	3223 (499)	2654 (184)	2087 (191)	3496 (527)

N.B: units no·m^{-2} (±SE); NS = Not Sampled

In 1971, the most abundant taxon in each area of Playgreen Lake was Amphipoda, followed by Mollusca and Ephemeroptera (Fig. 3.12). Diptera and Oligochaeta appeared less frequently than expected, possibly because the sorting techniques missed smaller forms. In 1987, the basic order of the 1971 abundances still held, with Amphipoda generally the most abundant taxon in each area, followed by Mollusca, Diptera, and Ephemeroptera. In 1989, an abrupt change occurred in the abundance ranking of taxa within each area, primarily because of a major decrease in the number of Amphipoda, but also because the relative abundance of Ephemeroptera increased.

In Kiskittogisu Lake, the 1989 standing stock in the south basin, which receives direct inflow from 8MC was 68% higher than in the more remote northern basin; this was in contrast to 1971 when the mean standing stocks in the two areas were almost identical (Fig. 3.10). As was the case with Playgreen Lake, the mean standing stock recorded in 1971 was low. The order of abundance of individual taxa in the northern portion of the lake in 1971 was Mollusca, followed by Ephemeroptera and then Amphipoda. Diptera and Oligochaeta were rare. In 1989 this order changed to Ephemeroptera, followed by Diptera, Amphipoda, and Mollusca (Fig. 3.13). The order of abundance in the southern portion in 1971 was Ephemeroptera, followed by Amphipoda and then Mollusca; this changed in 1989 to Ephemeroptera, followed by Mollusca and then Diptera.

Figure 3.11 Mean Abundance of Taxa Found in the Regions South and North of the 8-Mile Channel in Playgreen Lake, 1987 and 1989

Year	Region	Diptera	Amphipoda	Olig.	Mollusca	Ephem.	Trich.	Misc.	Total
1987	South	1250 (341)	5073 (1001)	355 (171)	3993 (839)	423 (98)	171 (27)	250 (51)	11516 (1298)
	North	1023 (85)	1998 (293)	15 (8)	1341 (175)	405 (93)	35 (8)	16 (51)	4833 (335)
1989	South	1410 (757)	513 (242)	341 (117)	1261 (339)	576 (116)	23 (8)	26 (7)	4151 (827)
	North	360 (41)	165 (54)	27 (11)	1061 (190)	995 (110)	6 (3)	16 (5)	2631 (159)

N.B: units no·m^{-2} (±SE); South = sites 1-17; North = sites 18-33, Olig. = Oligochaeta;
Mollusca: ~80% Pelecypoda and ~20% Gastropoda; Ephem. = Ephemeroptera, Trich. = Trichoptera

Figure 3.12 Mean Standing Stock of Individual Taxa in Areas 1 - 3 of Playgreen Lake, 1971, 1987, and 1989

	Year	Diptera	Amphipoda	Olig.	Mollusca	Ephem.	Trich.	Misc.
Area1	1971	15	201	1	170	98	12	6
	1987	1603 (499)	5586 (1459)	499 (255)	2989 (632)	437 (141)	199 (34)	245 (72)
	1989	1880 (1163)	658 (361)	385 (172)	896 (168)	553 (152)	27 (11)	17 (6)
Area 2	1971	0	141	0	251	33	11	1
	1987	829 (136)	3053 (759)	70 (40)	3883 (1391)	584 (130)	78 (28)	170 (52)
	1989	525 (65)	184 (104)	167 (79)	1298 (579)	1010 (200)	11 (6)	29 (11)
Area 3	1971	22	238	1	117	34	41	3
	1987	976 (95)	2186 (333)	8 (3)	1469 (209)	253 (54)	41 (9)	8 (5)
	1989	317 (28)	191 (71)	28 (15)	1299 (208)	793 (76)	7 (3)	19 (6)

N.B: units no·m^{-2} (±SE); South = sites 1-17; North = sites 18-33, Olig. = Oligochaeta; Ephem. = Ephemeroptera, Trich. = Trichoptera

Figure 3.13 Mean Standing Stock of Individual Taxa in Areas 4 and 5 of Kiskittogisu Lake, 1971 and 1989

	Year	Diptera	Amphipoda	Olig.	Mollusca	Ephem.	Trich.	Misc.
Area4	1971	8	49	2	85	55	0	7
	1989	502 (118)	330 (183)	49 (36)	259 (129)	916 (124)	0 (11)	32 (16)
Area 5	1971	8	68	0	36	97	0	0
	1989	431 (89)	139 (114)	135 (65)	1267 (232)	1503 (350)	4 (5)	16 (13)

N.B: units no·m^{-2} (±SE); South = sites 1-17; North = sites 18-33, Olig. = Oligochaeta; Ephem. = Ephemeroptera, Trich. = Trichoptera

The south basin of Playgreen Lake was the most productive and diverse region sampled in 1971, 1987 and 1989, probably because of the stabilizing influence of the large mass of water in Lake Winnipeg. Within the south basin, invertebrate populations were generally higher along the plume of suspended material entering Playgreen Lake from 2MC than in other parts of the basin. This indicates an enhancement of the benthic populations by the augmented flow from this channel, possibly by the addition of particulate organic material. Because the plume of sediment probably contains organic material washed from the shore of Lake Winnipeg, it provides a source of sustenance for the benthic invertebrates.

The portion of Playgreen Lake that lies north of 8MC appears to have suffered a decrease in the standing stock of benthic invertebrates following construction of the diversion into Kiskittogisu Lake. The significant diversion of water away from the northern areas of Playgreen Lake likely has removed nutrients and food that were formerly available to the benthos. After 8MC became operational, the effects of the added inflow to the south portion of Kiskittogisu Lake were particularly noticeable. With the longest water renewal time of any of the areas examined, the enhanced amounts of particulate organic matter caused an increase in abundance of invertebrates in the south portion over the north portion of the lake. The invertebrates in the south portion probably also benefitted from increased primary productivity caused by enhanced nutrients in the inflow.

The decline in total numbers of benthos in Playgreen Lake between 1987 and 1989 was probably caused by drought conditions across the prairie region in 1988-1989. Reduced spring and summer flows through the south basin of Playgreen Lake and warmer than normal air temperatures in northern Manitoba produced higher than normal water temperatures in the lake. Taxa which prefer cool lake waters, such as Amphipoda, declined, while taxa, such as Oligochaetes, known to thrive in warm shallow habitats benefitted by the increased temperatures and/or reduced flow.

WATERFOWL

Concerns expressed by the Norway House Band that opportunities for hunting waterfowl in the fall had declined after the regulation of Lake Winnipeg resulted in a FEMP study of waterfowl in this area. The purpose of this study was to investigate current waterfowl use in the fall and, to the extent, possible, compare it with historical usage.

One-on-one interviews were conducted with 7 waterfowl hunters in Norway House on February 13, 1987. Hunters were asked to share their observations on changes in the numbers and distribution of waterfowl; this information was used in the determination of the locations and timing of the 1987 surveys. A series of low-altitude aerial surveys were flown in 1986 and 1987, along the shorelines of Playgreen, Kiskitto and Kiskittogisu lakes and 8-Mile Channel (Fig. 3.14). The 1986 surveys were flown on September 5 and 12 in a Cessna 185 with two observers; the 1987 surveys were flown on October 5 ,16, and 26 in a Bell 206B helicopter with one observer. Surveys were flown at altitudes of between 30 and 40 m above the ground or water surface.

A summary of the 1986 and 1987 survey data is shown in Fig 3.15. Maximum densities are recorded because they were more likely to represent the potential for waterfowl use. Densities were consistently higher in 1987 than in 1986; this may be a reflection of the different survey techniques used each year, or it may indicate that

Figure 3.14 Waterfowl Survey Routes in the Outlet Lakes Area, 1986 and 1987

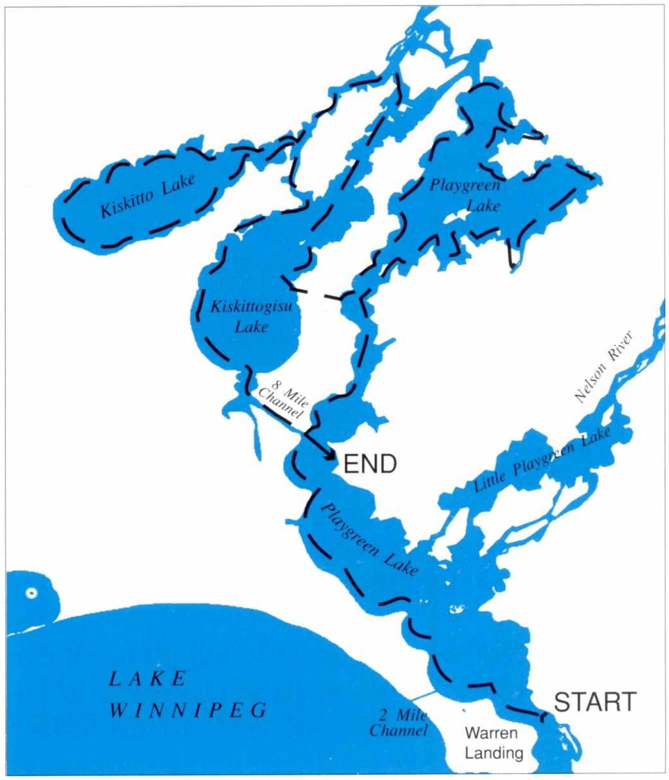


Figure 3.15 Waterfowl Densities[1] Observed during 1986/1987 Surveys of the Outlet Lakes Area

	Sep 5/86	Sep 12/86	Oct 5/87	Oct 16/87	Oct 26/87
Diving Ducks					
Playgreen-South Basin	0.00	0.07	0.67	1.46	2.37
Playgreen-North Basin	2.05	0.66	2.69	2.12	1.97
Kiskitto	0.12	0.00	0.00	0.13	N.S.
Kiskittogisu	0.25	0.33	1.04	0.00	0.00
Dabbling Ducks					
Playgreen-South Basin	0.03	0.00	1.19	3.06	0.24
Playgreen-North Basin	0.60	0.11	0.68	2.50	0.10
Kiskitto	0.00	0.00	0.30	0.00	N.S.
Kiskittogisu	0.30	0.00	1.36	4.58	0.00
Total Ducks[2]					
Playgreen-South Basin	0.03	0.07	2.26	4.86	2.37
Playgreen-North Basin	2.65	0.71	3.70	4.62	2.07
Kiskitto	0.12	0.00	0.30	0.13	N.S.
Kiskittogisu	0.55	0.33	2.40	4.58	0.00
Canada Geese					
Playgreen-South Basin	0.00	0.00	0.67	1.46	0.00
Playgreen-North Basin	0.10	0.19	0.11	0.16	0.00
Kiskitto	0.10	0.64	0.00	0.00	N.S.
Kiskittogisu	0.38	0.15	0.42	0.00	0.00

[1] Numbers shown are birds per kilometer based on survey lengths as follows: 396 km (Sept 5 and 12, 1986); 474 km (Oct 5, 1987); 333.3 km (Oct 16, 1987); and 292 km (Oct 26, 1987).
[2] Total ducks includes diving ducks, dabbling ducks, and unidentified ducks.
N.S.= not surveyed
X = highest density by area

the 1986 survey took place before large numbers of migrating waterfowl had reached the study area that year. There was considerable variation in waterfowl densities between surveys and sites, especially for dabblers (e.g. dabbler densities for the south basin of Playgreen Lake were 3.06 birds/km on October 16, 1987 and 0.24 birds/km for the same site on October 26, 1987).

The only historical data on the waterfowl use of the Outlet Lakes area are the results of two sets of aerial surveys conducted in 1972 and 1973 (LWCN 1975); these data are summarized in Fig. 3.16. As was the case with the 1986 and 1987 surveys, there was considerable variability in the waterfowl densities between surveys and sites in 1972 and 1973 (e.g. diver densities for the south basin of Playgreen Lake were 72.07 for September 7, 1973 and 14.21 for September 14, 1973).

Comparison of the 1972/1973 and 1986/1987 data suggest that diving ducks declined in the Outlet Lakes area by approximately 96%; declines were most apparent in the two basins of Playgreen Lake. It is risky, however,

Figure 3.16 Waterfowl Densities Observed during 1972/1973 Surveys of the Outlet Lakes Area

Diving Ducks	**Sep 24&25/72**	**Aug 30/73**	**Sept 7/73**	**Sept 14/73**	**Oct 6 /73**
Playgreen-South Basin	25.81	46.57	72.07	14.21	25.51
Playgreen-North Basin	63.15	102.01	N.S	8.19	N.S
Kiskitto	2.03	0.00	0.00	0.00	0.00
Kiskittogisu	4.83	1.88	0.00	0.00	3.24
Dabbling Ducks					
Playgreen-South Basin	1.49	0.00	0.11	0.00	0.85
Playgreen-North Basin	7.87	0.20	0.00	0.00	N.S
Kiskitto	0.90	0.00	0.00	0.77	0.00
Kiskittogisu	0.57	1.88	0.00	0.47	0.00
Total Ducks[2]					
Playgreen-South Basin	29.21	47.58	72.18	14.21	26.36
Playgreen-North Basin	71.02	104.22	N.S	8.51	N.S
Kiskitto	3.98	0.55	0.00	0.77	0.00
Kiskittogisu	8.39	3.22	0.00	0.47	3.24
Canada Geese					
Playgreen-South Basin	0.00	0.11	0.20	0.00	0.20
Playgreen-North Basin	0.78	0.44	N.S	0.14	N.S
Kiskitto	1.01	2.66	0.21	0.00	0.00
Kiskittogisu	1.13	0.27	0.22	0.51	0.45

[1]Numbers shown are birds per kilometer based on survey lengths as follows: 425.9 km (Sept 24 and 25, 1972); 220.2 km (Aug 30, 1973); 200.4 km (Sept. 7, 1973); 185.9 km (Sept 14, 1973); and 235.5 km (Oct 6, 1973)

[2]Total ducks includes diving ducks, dabbling ducks, and unidentified ducks.

N.S= not surveyed

X = highest density by area

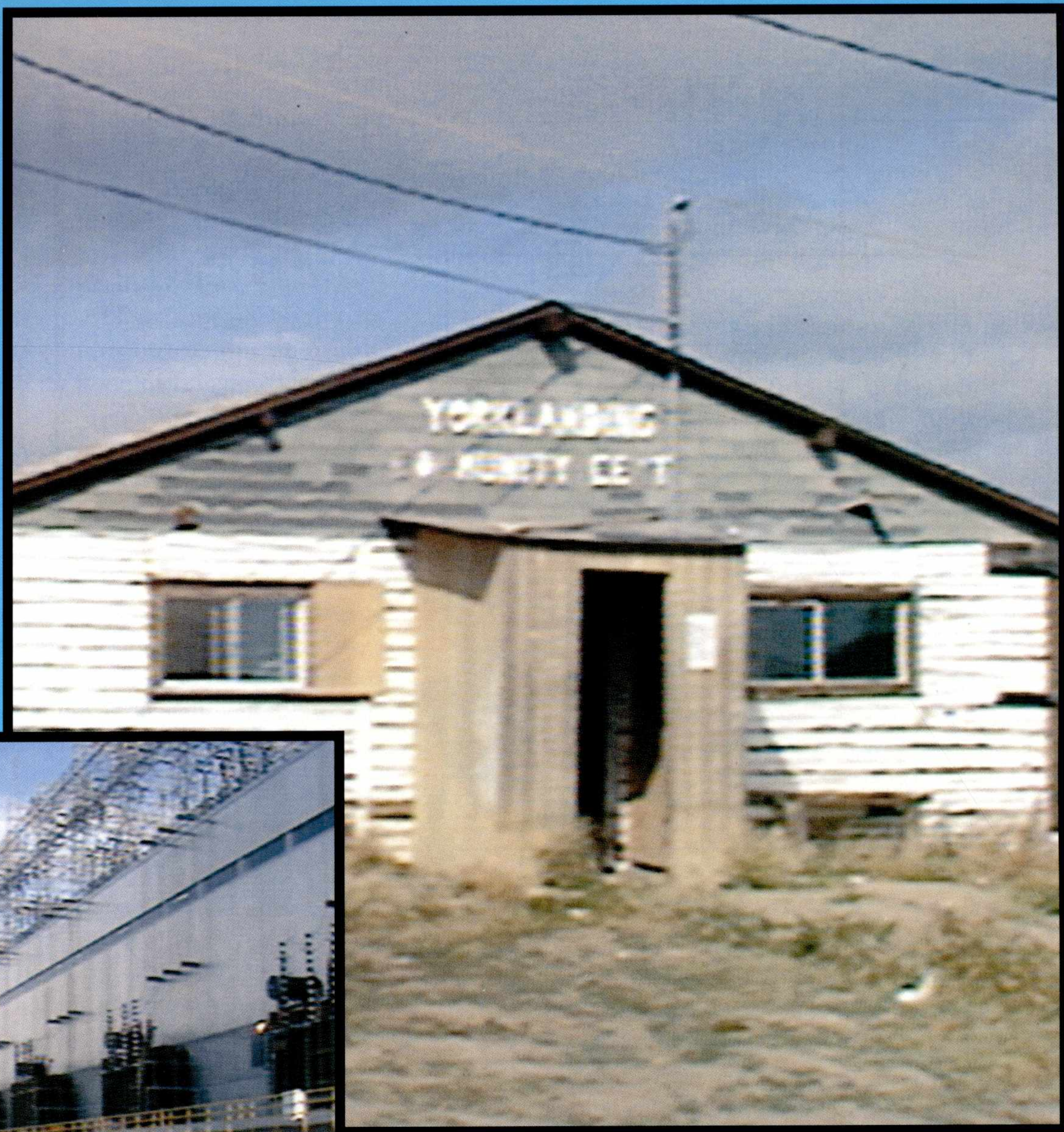
to draw broad conclusions about diver numbers on the basis of the 1972/1973 and 1986/1987 surveys, because both surveys provide only a limited number of "point-in-time" counts. Furthermore large between-year changes are not uncommon in waterfowl counts derived from aerial surveys. The data suggest that the Outlet Lakes area did not attract large numbers of dabbling ducks in either of the two survey periods. Similarly, only relatively small numbers of Canada geese were observed on individual surveys in both survey periods; indeed, no geese were observed on several surveys.

Two possible causes for the apparent change in diving ducks were considered: 1) changes in regional populations; and 2) modifications to the local physical environment and waterfowl habitat. The only data on regional populations are the annual breeding pair surveys conducted by the U.S. Fish and Wildlife Service (USF & WS) in northern Manitoba. These data provide adjusted population estimates for ducks and serve as general indicators of population size and trends. These data estimate the size of breeding populations; they do not indicate the size of fall migrating flocks.

Data from Stratum 24, the USF & WS region, which most closely approximate the FEMP study area, were reviewed for the periods 1964 - 1975 and 1977 - 1988. When the 12-year means for the two periods were compared, a small increase (approximately 2.6%) in diving ducks was found. However, large year-to-year fluctuations occurred during these periods; thus, the means on which these changes were based would vary somewhat as individual year counts were added or subtracted. The apparent reduction in diving duck use of the Outlet lakes area does not appear to be a reflection of a long-term decline in regional populations as represented by the Stratum 24 counts, since no such trends were evident in the stratum data.

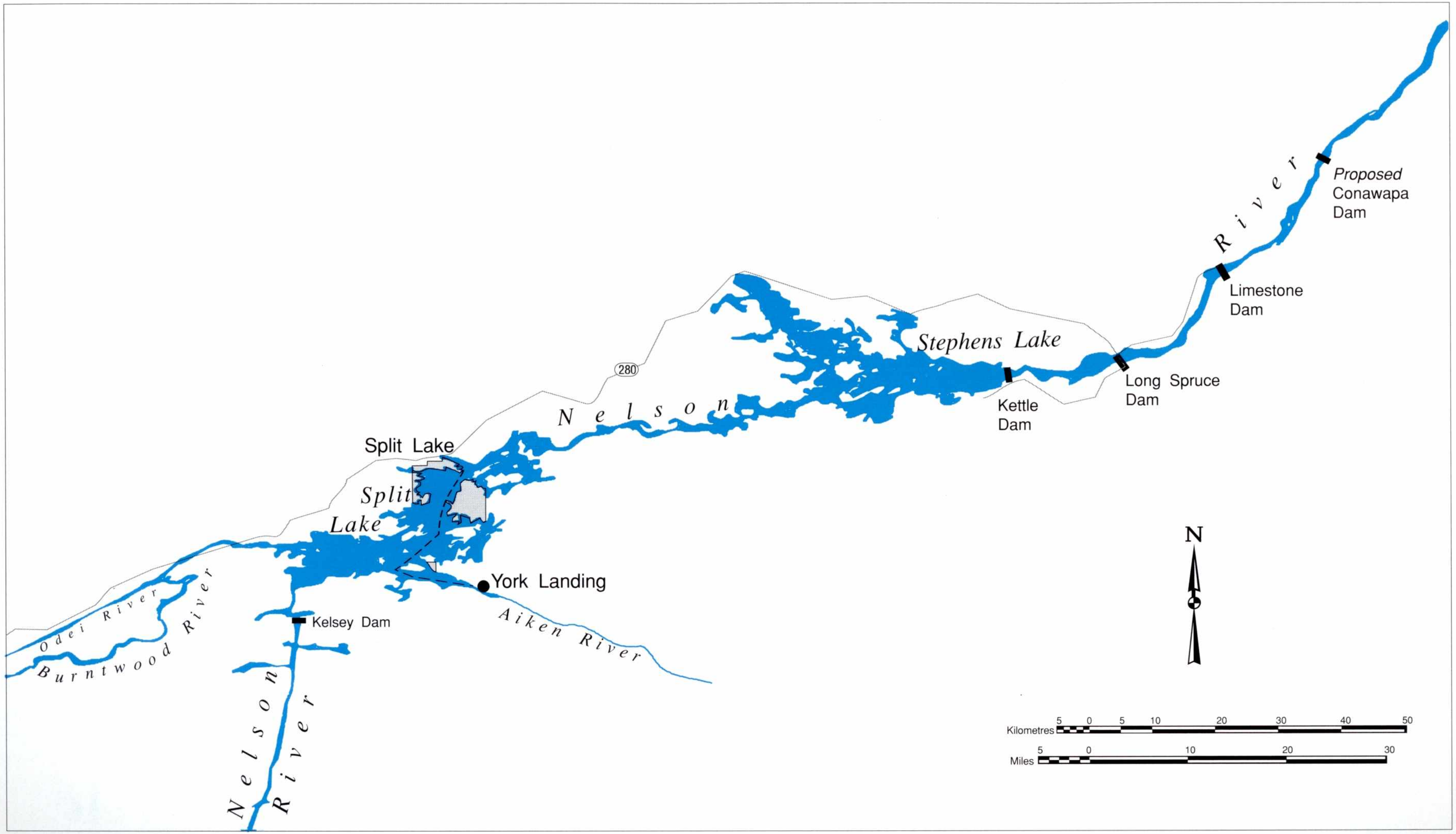
Modifications to the local physical environment and waterfowl habitat was the second possible cause considered for the apparent decrease in diving ducks. In particular, it was postulated that environmental changes might have caused a reduction in the availability of invertebrate food items in Playgreen Lake and/or a general reduction in the attractiveness of Playgreen Lake to diving ducks. No data are available on the abundance of benthic invertebrates in Playgreen Lake during the fall waterfowl staging period of September and October; consequently, no firm conclusions can be drawn regarding the abundance or availability of benthic invertebrates to fall staging diving ducks. Possible changes in environmental stimuli such as emergent vegetation, underwater visibility, and the appearance of the shoreline, may have reduced the potential of the habitat to attract diving ducks. Unfortunately, there is no direct experimental or survey proof to either validate or negate either of these postulated causes of the apparent decrease in diving ducks.

In summary, low numbers of ducks and Canada geese were observed in September 1986 and October 1987 in the Outlet Lakes area. Temporal and spatial data, as well as current data, were too limited to estimate changes in the waterfowl use of this area with any degree of confidence. The data suggest, however, that there may have been a decline in the diving duck use of this area following the regulation of Lake Winnipeg.



CHAPTER 4

SPLIT LAKE AREA



Proposed
Conawapa
Dam
Limestone
Dam
Long Spruce
Dam
Kettle
Dam
Stephens Lake
Nelson River
280
Split Lake
Split Lake
York Landing
Kelsey Dam
Aiken River
Odei River
Burntwood River
Nelson River
N
Kilometres
5 0 5 10 20 30 40 50
Miles
5 0 10 20 30

MORPHOLOGY AND SEDIMENT REGIMES

Introduction

The focus of the FEMP morphological study was the Churchill River Diversion (CRD) route, the region of the greatest morphological changes in the FEMP study area. In particular, a series of air photos was analyzed to determine the feasibility of air photo interpretation as a means of assessing past morphological changes along the CRD. Most of the results of this analysis were reported on in chapter 2; the results of the analysis for the First Rapids area are included in this chapter because of the proximity of this site to Split Lake.

Data on suspended sediment were collected at four sites in the Split Lake area as part of the FEMP water quality study; these data are summarized in this section. A brief discussion of suggested studies to better assess the changes in the morphology and sediment regime in the Split Lake area concludes this section.

Figure 4.1 Estimated Changes in Active Channel Area and Volume of River Erosion in the Vicinity of First Rapids

Surface area of river in	*Total (m^2)*	*Change in surface area river in*	*(m^2)*
1970	192,000	1970-1981	68,000
1981	260,000	1970-1986	80,000
1986	272,000	1981-1986	12,000

Estimated volume of eroded material (m^3)	low	high
1970-1981	98,000	441,000
1970-1986	111,000	499,500
1981-1986	13,000	58,500

Average annual sediment production (tonnes/year)	low	high
1977-1981	21,600	97,000
1981-1986	3,600	16,100

First Rapids

First Rapids is located approximately 14 km upstream of the confluence of the Burntwood River with the Odei River, in an area composed of glaciolacustrine basin deposits consisting of clay and silt, 2 to 15 m thick. Prior to the diversion, bedrock was exposed on both channel banks at First Rapids and the river channel was continuously confined by the valley walls. Downstream of First Rapids, the channel became much wider and the riverbanks consisted of high slopes composed of unconsolidated, likely lacustrine, sediments.

Inspection of pre-project photography indicated no detectable changes in channel morphometry had occurred in the vicinity of First Rapids in the period between 1950 and 1970. Comparative analyses of air photos indicated that the area of active channel at First Rapids increased from 22 000 m^2 in 1970 to 52 000 m^2 in 1981, with a further increase of 1 000 m^2 between 1981 and 1986. The area of active channel immediately downstream of First Rapids increased from 170 000 m^2 in 1970 to 208 000 m^2 in 1981 and 219 000 m^2 in 1985. Total area of active channel in the vicinity of First Rapids was 192 000 m^2 in 1970, 260 000 m^2 in 1981, and 272 000 m^2 in 1986 (Fig 4.1).

Assuming bank heights of 2 m and 9 m, for 'low' and 'high' estimates respectively, the amount of sediment eroded in the vicinity of First Rapids was estimated to be as low as 13 000 m^3 between 1981 and 1986 and as high as 499 500 m^3 between 1970 and 1986. Much of the shoreline erosion occurred during an ice jam in 1980. Average annual rate of sediment production was estimated to be between 2.2 to 9.7 x 10^4 tonnes per year from 1977 to 1981 and between 0.4 to 1.6 x 10^4 tonnes per year from 1981 to 1986.

Split Lake

Turbidity, total dissolved solids (TDS), non-filterable residue (NFR), and colour were sampled (Fig. 4.2) at 4 sites in the Split Lake area as part of the FEMP water quality study. Mean turbidity levels at all 4 sites, including the natural river site on the Aiken River exceeded the Canadian Water Quality Guidelines for turbidity in drinking water of 5 NTU. There were also occasions, at all 4 sites, when colour exceeded the Canadian Water Quality Guideline for colour in drinking water of 15 TCU.

Figure 4.2 Turbidity, Total Dissolved Solids, Non-Filterable Residue, and Colour for the Split Lake Area, 1986 - 1989

Variable	Type	Units	Mean	SD	Max.	Min.
Aiken River at the Inlet to Split Lake[1]						
Turbidity	field	NTU	7.[+]	3.	16.[+]	2.[+]
	lab	NTU	6.[+]	4.	18.[+]	1.
TDS		mg·L^{-1}	104.	51.	212.	27.
NFR	total	mg·L^{-1}	5.	4.	18.	<1.
	fixed	mg·L^{-1}	3.	2.	8.	<1.
Colour	true	relative	46.[+]	25.	120.[+]	8.
Burntwood River at the Inlet to Split Lake.						
Turbidity	field	NTU	34.[+]	11.	70[+,++]	17.[+]
	lab	NTU	35.[+]	10.	71.[+,++]	20.[+]
TDS		mg·L^{-1}	65.	3.	71.	58.
NFR	total	mg·L^{-1}	24.	16.	107.	10.
	fixed	mg·L^{-1}	21.	15.	100.	8.
Colour	true	relative	22.[+]	6.	40.[+]	10.
Nelson River below Grass River						
Turbidity	field	NTU	12.[+]	7.	45.[+]	4.
	lab	NTU	9.[+]	3.	18.[+]	5.
TDS		mg·L^{-1}	154.	21.	195.	119.
NFR	total	mg·L^{-1}	5.	2.	11.	2.
	fixed	mg·L^{-1}	4.	2.	9.	<1.
Colour	true	relative	9.	5.	20.[+]	<5.
Nelson River at the Outlet of Split Lake.						
Turbidity	field	NTU	22.[+]	12.	88.[+,++]	13.[+]
	lab	NTU	18.[+]	5.	35.[+]	11.[+]
TDS		mg·L^{-1}	118.	16.	154.	93.
NFR	total	mg·L^{-1}	10.	4.	21.	4.
	fixed	mg·L^{-1}	8.	3.	19.	3.
Colour	true	relative	14.	6.	30.[+]	5.

[1] Sample period was Jan 1987 to Oct 1989

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

Mean turbidity and NFR values at the Burntwood River inlet to Split Lake were the highest of the 4 FEMP sample sites in the Split Lake area. These higher values were likely the result of the documented bank erosion near First Rapids. Mean turbidity levels for the Burntwood River at the inlet to Split Lake were almost 4 times more turbid than the water flowing into Split Lake from the Nelson River, while turbidity levels at the outlet of Split Lake were intermediate between the two upstream values.

Lack of pre-development data for the Burntwood River inlet to Split Lake prevented the identification of changes in turbidity or NFR at this site. However, analysis of the data record for upstream sites at Threepoint Lake and Thompson, as well the documented bank erosion along the CRD, suggest that turbidity likely increased at the Burntwood River inlet site. There was a 67% decline in NFR at the Nelson River site at the inlet to Split Lake. As discussed in the water quality section, water quality changes observed at this site appear to be restricted to the reach between Split and Sipiwesk lakes.

Comparison of the FEMP data collected at the outlet of Split Lake with pre-development data showed no change in mean turbidity levels (1987 - 1989: 18 NTU; 1972 - 1973: 17 NTU) or mean NFR levels (1987- 1989: 8 mg·L^{-1}; 1972 -1973: 6 mg·L^{-1}). Although NFR in the Burntwood River is significantly higher than in the Nelson River, a large amount of sediment, most especially from the lower diversion area, appears to have deposited in Split Lake. No studies of sedimentation rates have been conducted in Split Lake; however, on the basis of turbidity measurements and analyses of satellite images, it appears that most of the sediment has been deposited near the Burntwood River inlet forming a subaqueous delta.

A belief among Split Lake residents that the water is more turbid now than in the past was not substantiated by the results of the FEMP study; however, there are two possible explanations for this apparent difference. The limited spatial coverage of Split Lake in the FEMP study may have been too insensitive to detect localized changes in areas of particular interest to community residents, such as in the community's beach area, or alternatively Kelsey Dam might have caused an increase in turbidity immediately following its installation in the early 1960s, followed by a subsequent decrease as the Kelsey Dam forebay area stabilized. Any decrease in turbidity in Split Lake from the Nelson River would have been compensated for by increases from the Burntwood River, resulting in no detectable change during the sampling period. The virtual absence of pre-development data for the Kelsey Dam, and the paucity of data for the immediate post-impoundment period, make it difficult to examine this possibility. Unless some other data can be found and validated, it may not be possible to completely document the effects the LWCN project on the sediment regime of Split Lake.

Future Studies

For the lower CRD area, from First Rapids to Split Lake, the same basic morphological monitoring program is recommended as that for the upper CRD area, in the Threepoint and Footprint lakes area, namely: 1) to undertake field reconnaissance to ground truth the air photo interpretation; 2) to prepare planimetric mapping and classify shoreline characteristics; and 3) to obtain local participation in developing a classification system as relevant to the northern residents as possible. In addition, the re-survey of some river cross sections and the establishment of at least 2 sounding lines in Split Lake, immediately off the mouth of the Burntwood River, are recommended. In particular, the rate and spatial extent of sediment deposition in Split Lake, off the mouth of the Burntwood River should be determined, by using such methods as bathymetric surveys, sediment coring, and sedimentation pans.

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; 4 of these 11 sites were located in the Split Lake area. This section presents the results of the water quality study for this 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 exists; 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 the NFC communities.

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

One of the 3 natural sites monitored in the FEMP water quality study was in the Split Lake area, on the Aiken River where it enters Split Lake (Fig. 4.3). These 3 sites were monitored 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.

As was the case for the other 2 natural rivers, the water quality of the Aiken River was characterized by a number of variables which exceeded the Canadian Water Quality Guidelines. Mean values for turbidity, colour, extractable aluminum (AL), total copper (Cu) and extractable iron (Fe) all exceeded the Guidelines and there were sporadic occurrences when extractable manganese (Mn) exceeded the Guidelines (Figs. 4.2 + 4.4).

Due to the lack of pre-development water quality data and of any discharge data for the Aiken River, no analyses of temporal trends or of water quality - discharge relationships were possible for this site.

Distinct differences between the ice-free and ice-cover seasons were evident in 9 variables (Fig. 4.5). Alkalinity (total and bicarbonate), hardness, dissolved calcium (Ca), total ammonia (NH_3), dissolved inorganic carbon (DIC), and dissolved arsenic (As) and zinc (Zn) were higher under ice-cover, while chlorophyll a was lower. This is generally the same pattern as in the Footprint River natural site, suggesting a common cause.

Figure 4.3 FEMP Water Quality Sites in the Split Lake Area

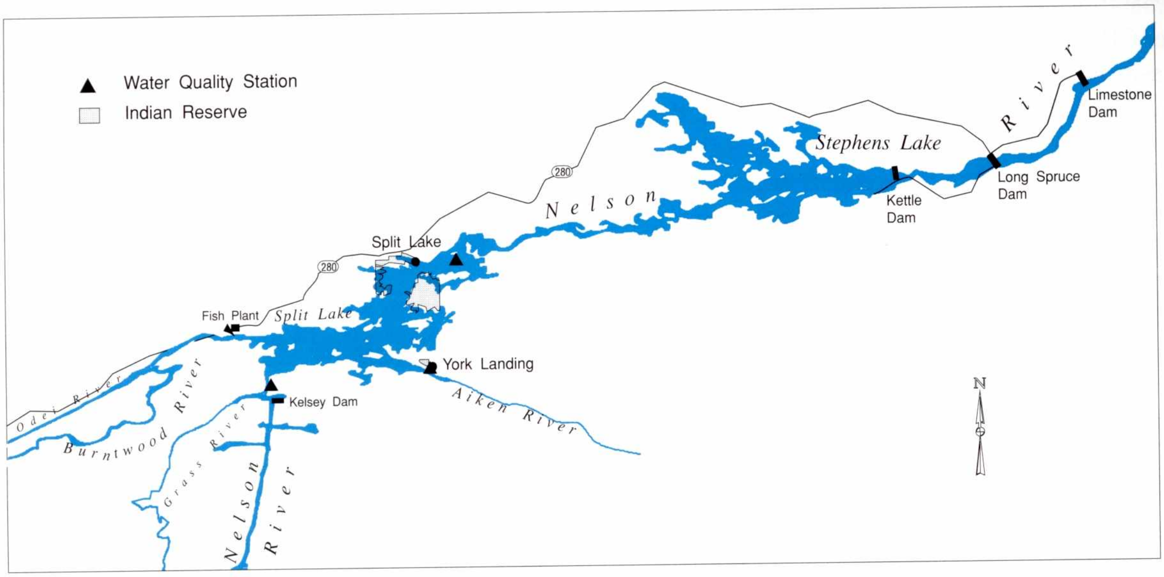
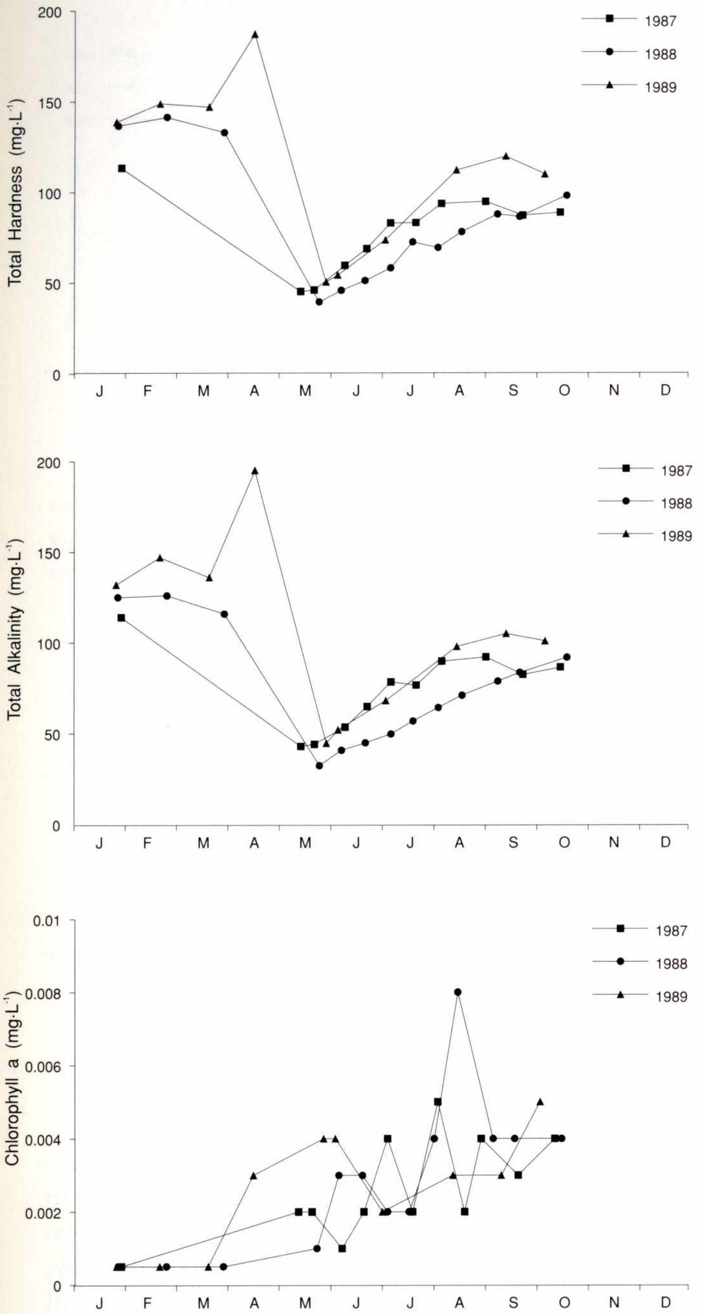


Figure 4.4 Water Quality Variables Which Exceeded the Canadian Guidelines for the Split Lake Area, 1986 - 1989

Variable	Type	Units	Mean	SD	Max.	Min.
Aiken River at the Inlet to Split Lake, Jan 1987 - Oct 1989						
Al	extractable	$mg \cdot L^{-1}$	0.107++	0.059	0.226++	<0.100
Cu	total	$mg \cdot L^{-1}$	0.0029++	0.0041	0.0170++	<0.0005
Fe	extractable	$mg \cdot L^{-1}$	0.438+,++	1.008	5.980+,++	0.101
Mn	extractable	$mg \cdot L^{-1}$	0.093+	0.339	1.920+	0.005
Burntwood River at the Inlet to Split Lake, Sept 1986 - Oct 1989						
Al	extractable	$mg \cdot L^{-1}$	0.592++	0.315	2.140++	0.235++
Cd	total	$mg \cdot L^{-1}$	<0.0001	0.0001	0.0003++	<0.0001
Cu	total	$mg \cdot L^{-1}$	0.0048++	0.0062	0.0278++	0.0010
Fe	extractable	$mg \cdot L^{-1}$	0.774+,++	0.416	2.860+,++	0.299
Pb	total	$mg \cdot L^{-1}$	0.0010	0.0007	0.0034++	<0.0007
Mn	extractable	$mg \cdot L^{-1}$	0.030	0.027	0.177+	0.004
Nelson River below Grass River, Sept 1986 - Oct 1989						
Al	extractable	$mg \cdot L^{-1}$	0.188++	0.101	0.418++	<0.100
Cu	total	$mg \cdot L^{-1}$	0.0037++	0.0043	0.0159++	0.0010
Fe	extractable	$mg \cdot L^{-1}$	0.196	0.074	0.409++	0.045
Nelson River at the Outlet of Split Lake, Sept 1986 - Oct 1989						
Al	extractable	$mg \cdot L^{-1}$	0.315++	0.126	0.651++	<0.100
Cu	total	$mg \cdot L^{-1}$	0.0050++	0.0075	0.0308++	0.0010
Fe	extractable	$mg \cdot L^{-1}$	0.389+,++	0.110	0.672+,++	0.129
Pb	total	$mg \cdot L^{-1}$	0.0009	0.0011	0.0066++	<0.0007

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

Figure 4.5 Seasonal Water Quality Trends for Selected Water Quality Variables in the Aiken River



Split Lake Inlets

Two FEMP sample sites were located on the two major inlets to Split Lake: the Burntwood River and the Nelson River (Fig. 4.3). Water quality at the Burntwood River inlet to Split Lake, during 1987 - 1989, was typical for the Burntwood River below the confluence with the Churchill River diversion. Concentrations of all but two variables were the same as at the Thompson site; the exceptions were the 45 - 50% higher concentrations for the annual mean total and fixed non-filterable residue (NFR). Mean levels for turbidity, colour, extractable Al, total Cu, and extractable Fe all exceeded the Canadian Water Quality guidelines; there were sporadic occasions when total cadmium (Cd), lead (Pb) and extractable manganese (Mn) exceeded the guidelines (Figs. 4.2 + 4.4).

Physical and chemical water quality in the Nelson River at the inlet to Split Lake, during 1987 - 1989, was not significantly different in any aspect from that at Sea River Falls, upstream. Mean levels of turbidity, extractable Al and total Cu all exceeded the Canadian Water Quality Guidelines; there were sporadic occasions when colour and extractable Fe exceeded the guidelines (Figs. 4.2 + 4.4).

No quantitative analysis of changes in water quality since the Churchill River diversion is possible for the Burntwood River inlet site due to lack of pre-development data; however, it is likely that changes in water quality at this site were similar to those documented upstream at Thompson. Several significant differences were noted for the Nelson River inlet site between the 1971 - 1974 pre-development period and the 1987 - 1989 FEMP period (Fig. 4.6) (e.g. declines in NFR, total dissolved phosphate (TDP) and total dissolved nitrogen (TDN) of 67, 45 and 22% respectively). These changes appear to have been restricted to the reach between Sipiwesk and Split lakes. Some of these changes might be explained by a natural evolution of limnological conditions in the Kelsey Generating Station forebay or alternatively they may be due to the temporary decrease in erosion rates as a result of the lower Nelson River flows. Between 1987 and 1989 Nelson River flows were at or near the lowest observed in the past 30 years. Additional sampling, in periods of non-drought conditions, is required to properly determine if the LWCN project has had any long-term impact on water quality at this site.

Figure 4.6 Changes in Selected Water Quality Variables in the Nelson River near the Inlet of Split Lake from 1972/1973 to 1987/1989

	NFR	Diss Ca	TDP	TDN	Diss Fe
	NTU	$mg{\cdot}L^{-1}$	$mg{\cdot}L^{-1}$	$mg{\cdot}L^{-1}$	$mg{\cdot}L^{-1}$
X_{pre}	15.0	28.5	0.020	0.388	0.113
X_{post}	5.0	29.3	0.011	0.304	0.039

X_{pre}=1972/1973 mean; x_{post} = 1987/1989 mean

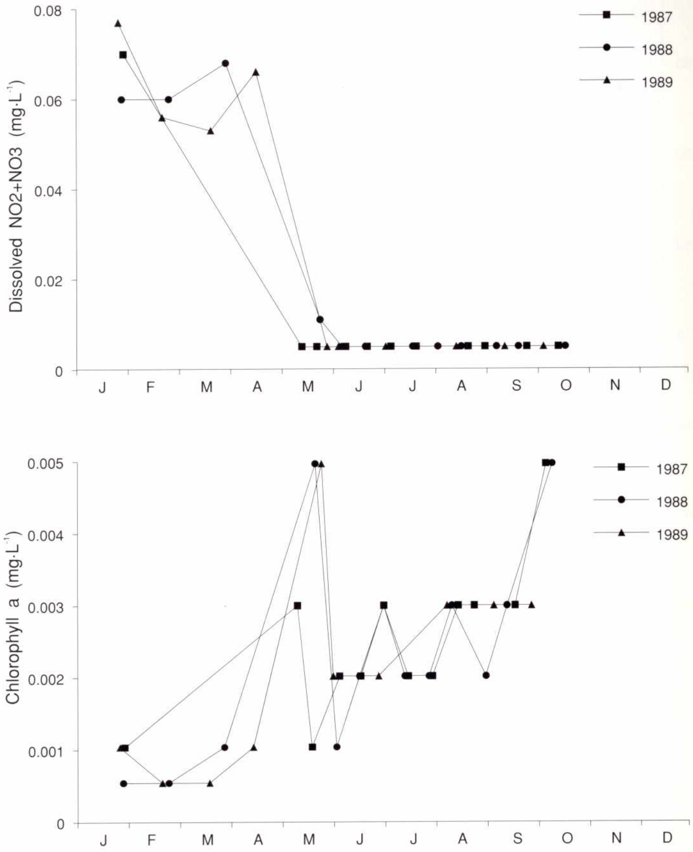
Positive correlations with discharge dominated at the Burntwood River inlet site during 1987 - 1989, consistent with the FEMP findings at sites upstream on the Burntwood River. The positive correlates, which included turbidity, are evidence of the increased importance of eroded shoreline materials in regulating water quality. However, none of the relationships was strong enough to show much impact prediction or management potential.

Fewer variables were significantly correlated with discharge at the Nelson River inlet site than at sites in the Burntwood River drainage or upstream on the Nelson River. Most of the significant correlations had positive slopes, but none of the significant correlations were particularly strong. None of the significant correlations with discharge for the 1987 - 1989 period resembled any which existed during the pre-development (1972 - 1973) period. The pre-development discharge relationships at this site followed the trends evident in the natural rivers and in the Burntwood River at Thompson prior to development, with negative correlates dominating and positive correlations restricted to variables associated with particulate matter or suspended sediments. The cause of these changes is less clear than was the case at sites on the Burntwood River. Additional sampling during periods of higher discharge is required to determine if the relationships observed during the FEMP study are representative of current conditions or if some other relationships represent the usual post-development condition.

Concentrations of dissolved NO_2-NO_3 at the Burntwood River inlet site were higher under ice-cover, while chlorophyll a levels were lower under ice-cover than in the open-water season (Fig. 4.7). The causes of these seasonal patterns, which were also observed at sites upstream on the Burntwood River, were probably the same as at the upstream sites, which were discussed in chapter 2.

Few variables at the Nelson River inlet site showed marked differences between the open-water and ice-cover seasons during 1987 - 1989, although conductivity, hardness, dissolved magnesium (Mg) and dissolved NO_2-NO_3 tended to be higher under ice cover (Fig. 4.8). The pre-development database for this site is insufficient to conduct a quantitative analysis of the differences between the ice-free and ice-cover seasons; however, a qualitative analysis suggests that the seasonal patterns of several variables were somewhat different during the pre-development period than during the

Figure 4.7 Seasonal Water Quality Trends for Selected Water Quality Variables in the Burntwood River at the Inlet to Split Lake

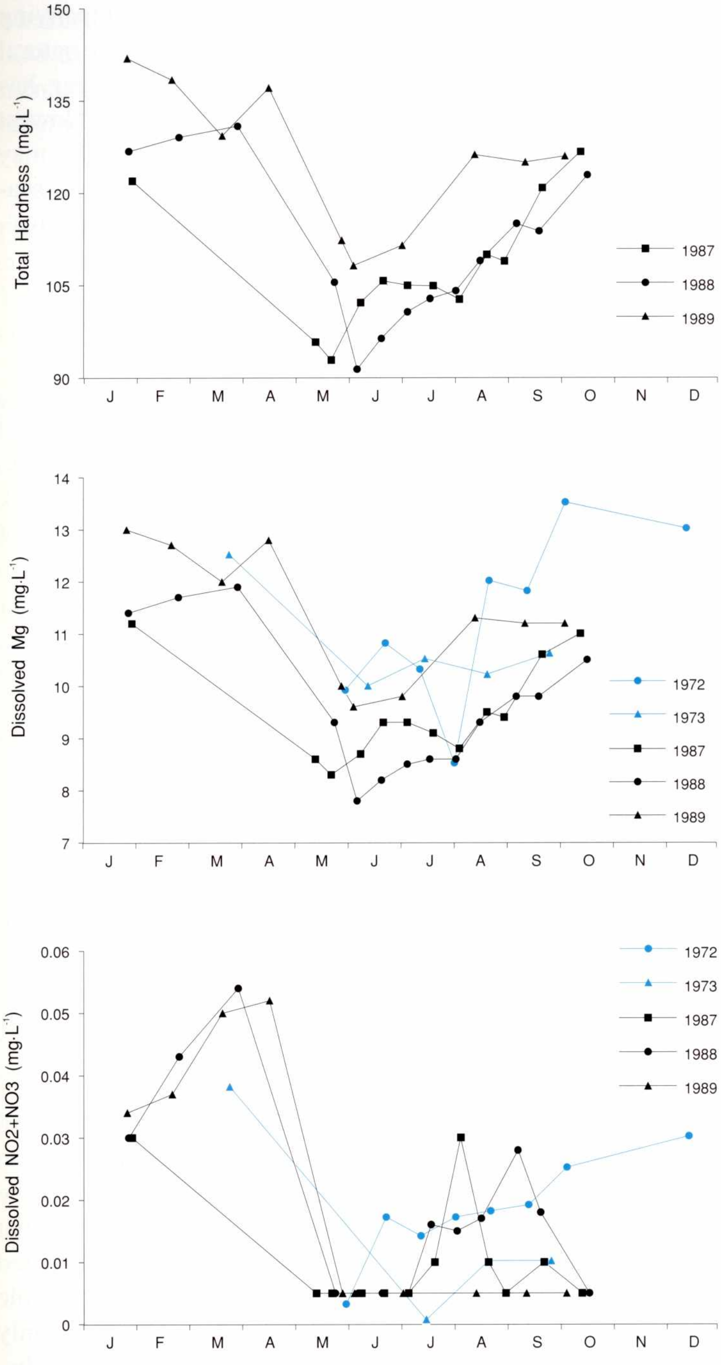


FEMP study. As noted previously, low Nelson River flows during the FEMP study may have been responsible for the observed values.

Split Lake Outlet

Water quality at the outlet of Split Lake, from 1987 to 1989, shared a number of general similarities with sites upstream on the Nelson River. Mean levels of turbidity, extractable Al and total Cu exceeded the guidelines at all the Nelson River sites (Figs. 4.2 + 4.4); however water quality at the outlet of Split Lake differed from the other upper Nelson River sites in the magnitude by which these guidelines were exceeded. For example, turbidity and extractable Al at the outlet of Split Lake exceeded the guidelines by factors of 4 and 3.2 respectively, but only by factors of 2 on average at the other upper Nelson River sites. Another difference between the Split Lake outlet and other upper Nelson River sites was the greater

Figure 4.8 Seasonal Water Quality Trends for Selected Water Quality Variables in the Nelson River at the Inlet to Split Lake



concentration of extractable Fe at the Split Lake outlet; the higher levels appear to be a direct result of the Churchill River diversion. Other characteristics of the water quality at the Split Lake outlet include the sporadic occurrences of total lead (Pb) and colour at levels above the guidelines.

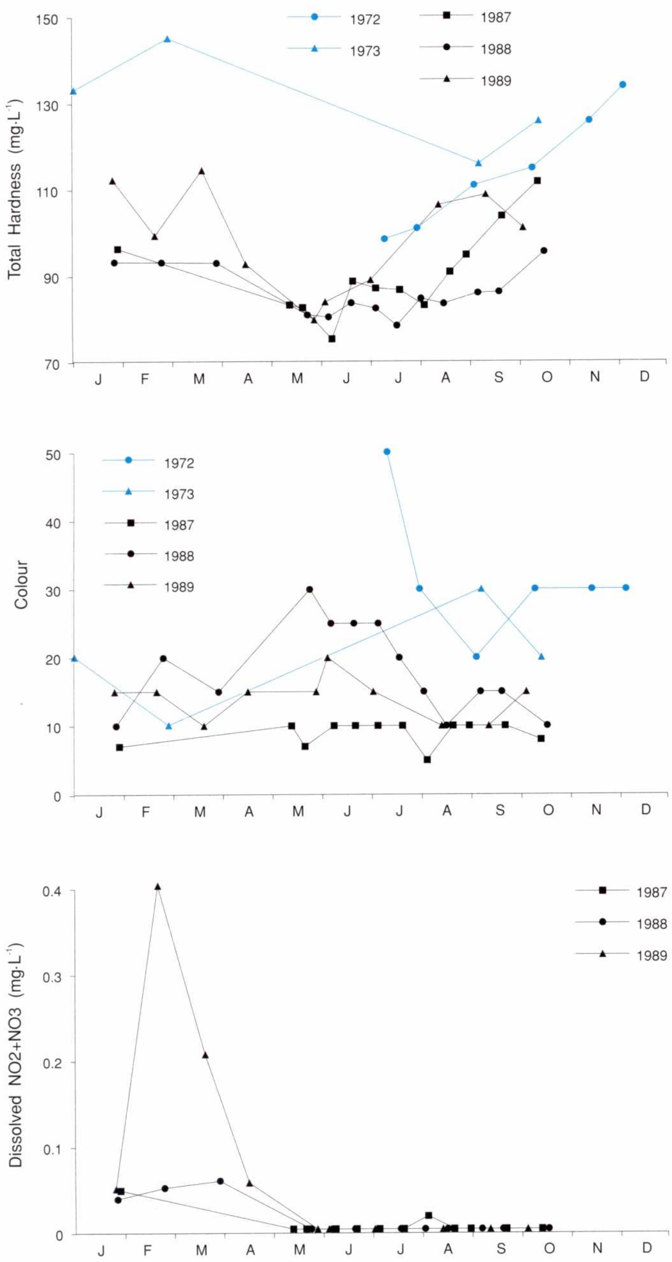
Statistical analysis of pre-development data (1972 - 1973) and post-development data revealed significant declines had occurred in a number of variables (e.g. conductivity, hardness, major ions, etc., (Fig. 4.6); most of the changes were solely attributable to the increased flows of softer water from the Churchill River drainage basin into Split Lake as a result of the diversion. An increase in extractable Fe, and significant decreases in total dissolved solids (TDS), dissolved inorganic carbon (DIC) and dissolved arsenic (As) were inferred on the bases of current differences in concentration between the Burntwood and Nelson rivers and the relative contribution of these rivers to total inflow.

Analysis of the FEMP data does not indicate that the Churchill River diversion has caused a significant increase in annual mean water turbidity (1972 - 1973 mean, 17 NTU; 1987 - 1989 mean, 18 NTU) or suspended sediment concentration (NFR fixed: 1972 - 1973 mean, 6 $mg{\cdot}L^{-1}$; 1987 - 1989 mean, 8 $mg{\cdot}L^{-1}$) or a decrease in water transparency, despite the much higher suspended sediment concentrations in the Burntwood River than in the Nelson River. Much of the suspended sediment load of the Burntwood River appears to settle out near the mouth, preventing a general increase in suspended sediment levels in the lake or downstream.

Only 3 variables were significantly correlated with discharge in the 1987 - 1989 study period; all correlations were positive and were not particularly strong. There were 10 water quality - discharge relationships for the pre-development period, based on data for two sites near the outlet of Split Lake; all correlations were negative and fairly strong. The shift from negative to positive correlations at the Split Lake outlet, although consistent with the changes in the discharge relationships in the Burntwood River following diversion, cannot be conclusively attributed to the diversion. Similar changes in the discharge relationships in the Nelson River at the inlet to Split Lake suggest a possible influence of the low Nelson River flows during the FEMP study or possibly some impact of the LWCN project on the upper Nelson River and Lake Winnipeg. Additional sampling during a period of higher Nelson River flows is required to determine the cause of the change in the discharge relationships at the outlet of Split Lake.

In contrast to the pre-development period, few variables occurred in different concentrations under ice-cover than in the open-water season during the FEMP study and none was common to the pre- and post-development periods (Fig. 4.9). Those which showed some evidence of seasonal variation after development (e.g. dissolved NO_2-NO_3) occurred in higher concentrations under ice-cover; this occurrence is generally consistent with the higher levels in both major inflows during the ice-cover period. The absence of seasonal differences in several variables during the FEMP study could be due to one or more of the several changes in the water regime which have occurred. For example the low Nelson River flows during the FEMP study period may have disrupted the normal discharge relationships and accompanying seasonal trends. Alternatively, the reversal of the natural seasonal flow regime, due to the operation of Jenpeg, has diminished the difference between the highest and lowest monthly mean flows at the outlet of Split Lake; this may have been sufficient to eliminate the rather small seasonal differences that prevailed prior to development.

Figure 4.9 Seasonal Water Quality Trends for Selected Water Quality Variables in the Nelson River at the Outlet of Split Lake



Bacteriological Assessment

A bacteriological assessment of the recreational waters in the Split Lake and York Landing areas was conducted in 1987. Seasonal studies were conducted from May to September at 5 sites at Split Lake and 4 sites at York Landing; in addition, an intensive study in the vicinity of the sewage treatment plant at Split Lake was conducted in August. The geometric mean fecal coliform bacteria for the seasonal studies ranged between 6 and 37 per 100 mL at Split Lake and 1 and 10 per 100 mL at York Landing. The geometric mean for the intensive study was 21 per 100 mL. Comparison of the results of the intensive study to the Health and Welfare Canada Recreational Water Quality Guideline 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 studies precludes direct comparison of their results to this guideline.

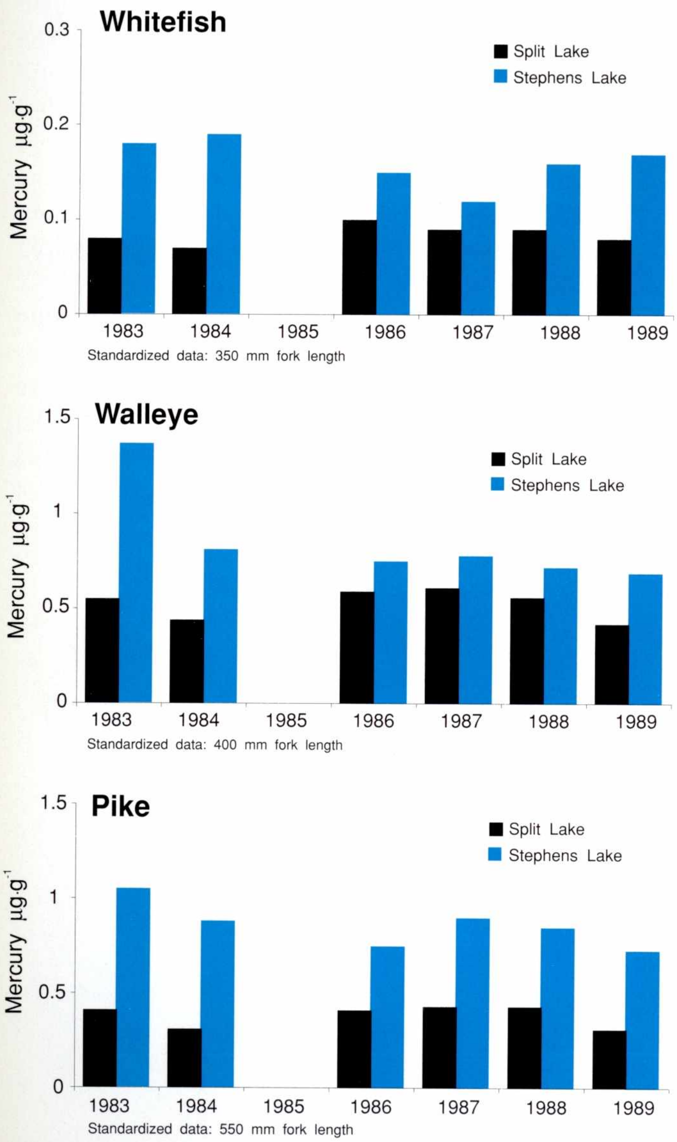
MERCURY

The FEMP mercury program focused on the Churchill River Diversion (CRD) route since this was the region within the FEMP study area that has experienced the most severe mercury problem. The basic processes that govern the production of methyl mercury, the expected longevity of elevated mercury levels in fish, and possible mitigative measures were discussed in chapter 2. Only mercury issues specific to the Split Lake area are discussed in this chapter, namely: 1) current mercury levels in fish in Split and Stephens lakes; 2) current mercury levels in residents of Split Lake and York Landing; and 3) a discussion of the possible factors affecting mercury levels in fish in Split Lake.

Mercury levels in fish in Split and Stephens lakes were sampled by the Manitoba Department of Natural Resources (Fig. 4.10). Whitefish mercury levels in Split Lake were within the average for northern Manitoba lakes remote from hydro development (less than 0.1 $\mu g \cdot g^{-1}$). Mercury levels in walleye from Split Lake ranged from 0.42 to 0.61 $\mu g \cdot g^{-1}$ during the period 1983-1989, while mercury levels in pike from Split Lake were slightly lower, ranging from 0.31 to 0.41 $\mu g \cdot g^{-1}$ during this period.

Whitefish mercury levels in Stephens Lake were approximately double those of Split Lake. Walleye mercury levels in Stephens Lake declined significantly from 1.37 $\mu g \cdot g^{-1}$ in 1983 to 0.8 $\mu g \cdot g^{-1}$ in 1984 and stabilized near this lower level from 1984 to 1989. As late as 1989,

Figure 4.10 Standardized Mean Muscle Mercury Concentrations for Fish in Split and Stephens Lakes, 1983 - 1989



18 years after impoundment, mercury levels in walleye were 3 times higher than the pre-development level of 0.25 $\mu g \cdot g^{-1}$ reported for Moose Lake, which was inundated in the formation of Stephens Lake. Pike mercury levels in Stephens Lake also declined significantly from 1.05 to 0.75 $\mu g \cdot g^{-1}$ between 1983 and 1986 and remained near this lower level through 1989. In 1989, mercury levels were 4 times the pre-development concentration of 0.20 $\mu g \cdot g^{-1}$ reported for Moose Lake.

Mercury levels in residents of Split Lake and York Landing were monitored by Health and Welfare Canada (Fig. 4.11). Of the approximately 15% of the population of Split Lake tested in 1989/90, 98% had values in the range of 0-19 ppb, which is considered to be in the "normal" range: the remaining values were in the range of 20-50 ppb. All of the women of childbearing age who were tested had values in the 0-19 ppb range.

Of the approximately 33% of the population of York Landing tested in 1988/89, 98% had values in the range of 0-19 ppb, with the remaining values being in the range of 20-50 ppb. All of the women of childbearing age who were sampled had values in the 0-19 ppb range. Unlike South Indian Lake and Nelson House, there were never any mercury levels in the tested populations of Split Lake and York Landing that exceeded 99 ppb.

Because the mercury levels in walleye and pike in Split Lake exceed 0.2 $\mu g \cdot g^{-1}$, residents of Split Lake and York Landing, with mercury levels in excess of 20 ppb, have been advised by Health and Welfare Canada to avoid the consumption of these species as a staple in their diet. This restriction has caused concern in the communities and raised questions regarding the effect of past hydro development on fish mercury levels in the lake.

While mercury levels in pike and walleye from Split Lake are not unusual in comparison to the natural range in northern Manitoba, they do appear to be uncharacteristically high for non-flooded lakes in the FEMP study area. The seemingly high mercury levels in walleye and pike are not related to the diversion of the Churchill River as they were prevalent before the diversion began operating (Fig. 4.12). Furthermore, as noted in chapter 2, measurements of methyl mercury in water in 1989 indicated methyl mercury produced in Notigi Reservoir does not travel as far as Thompson, and therefore could not be affecting fish mercury levels in Split Lake, farther downstream.