Effects of oil sands related aquatic reclamation on yellow perch (*Perca flavescens*). I. Water quality characteristics and yellow perch physiological and population responses

M.R. van den Heuvel, M. Power, M.D. MacKinnon, T. Van Meer, E.P. Dobson, and D.G. Dixon

Abstract: In order to test the viability of oil sands aquatic reclamation techniques, yellow perch (*Perca flavescens*) were stocked into three experimental ponds. Pond substrates consisted of either oil sands fine tailings or clay and lean oil sands deposited by the mining operations. Yellow perch were stocked immediately postspawning and subsamples were sacrificed at 5 and 11 months to measure indicators of energy storage and utilization. These indicators included survival, age, spawning periodicity, condition factor, gonad size, fecundity, and liver size. Indicators generally showed patterns consistent with improved energy storage and utilization in the experimental pond yellow perch as compared with yellow perch in the lake from which the stocked fish originated. This was evidenced by increased gonad size, condition factor, and liver size and the disappearance of spawning periodicity. The patterns observed in experimental ponds suggest improved resource availability and (or) reduced intra- and interspecific competition. Yellow perch physiological indicators were also compared with those measured at several remote natural lakes in the area. Fisheries parameters measured in yellow perch from the experimental ponds generally fell within the range of those found in natural lakes.

Résumé : Pour vérifier la viabilité des techniques de récupération des sables bitumineux en milieu aquatique, on a introduit des perchaudes (*Perca flavescens*) dans trois étangs expérimentaux. Les substrats des étangs étaient constitués de résidus fins de sables bitumineux, ou d'argile et de sables bitumineux pauvres issus des opérations minières. Les perchaudes ont été introduites immédiatement après la fraye, et on a sacrifié des sous-échantillons à 5 et 11 mois pour mesurer les indicateurs suivants de stockage et d'utilisation de l'énergie : taux de survie, âge, périodicité de la fraye, coefficient de condition, taille des gonades, fécondité et taille du foie. Les indicateurs ont de façon générale montré une amélioration du stockage et de l'utilisation de l'énergie chez les perchaudes des étangs expérimentaux par rapport aux perchaudes du lac d'origine de ces perchaudes introduites. On a en effet observé un accroissement de la taille des gonades, du coefficient de condition et de la taille du foie et la disparition de la périodicité de la fraye. Les profils observés dans les étangs expérimentaux laissent croire qu'il y avait dans ces milieux davantage de ressources disponibles et (ou) une réduction de la compétition intraspécifique et interspécifique. On a aussi comparé les indicateurs physiologiques des perchaudes avec ceux mesurés dans plusieurs lacs naturels éloignés de la région. Les valeurs des paramètres halieutiques mesurés chez les perchaudes des étangs expérimentaux étaient généralement comparables à celles observées dans les lacs naturels.

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Introduction

Oil sands deposits in northeastern Alberta, Canada, are a vast reservoir of fossil fuel estimated to contain one third of the world's known recoverable oil. Current levels of production from the Athabasca oil sands deposit represent 20% of Canada's petroleum needs. This level is certain to increase as conventional oil reserves dwindle and oil sands mining and extraction technologies improve.

The hot water flotation method for extraction of heavy hydrocarbons (bitumen) from oil sands produces large volumes of process water containing a suspension of solids as well as un-

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M.R. van den Heuvel,[1] E.P. Dobson, and D.G. Dixon. Department of Biology, University of Waterloo, Waterloo, ON N2L 3G1, Canada.
M. Power. Department of Agricultural Economics, University of Manitoba, Winnipeg, MB R3T 2N2, Canada.
M.D. MacKinnon. Syncrude Canada Ltd., Edmonton Research Center, 9421 17th Ave., Edmonton, AB T6N 1H4, Canada.
T. Van Meer. Syncrude Canada Ltd., Environment Department, P.O. Bag 4009, M.D. 0078, Fort McMurray, AB T9H 3L1, Canada.

[1]Author to whom all correspondence should be addressed. Present address: New Zealand Forest Research Institute, Private Bag 3020, Rotorua, New Zealand. e-mail: mike.vandenheuvel@forestresearch.co.nz

recovered hydrocarbons. These tailings segregate into deposits of sand and a slowly densifying suspension of water, fine clay particles (<22 μm), and unrecovered bitumen that is referred to as fine tailings. The oil sands fine tailings release pore water while densifying into mature fine tailings, a stable aqueous suspension containing 65% water (Fine Tailings Fundamentals Consortium 1995). The enormous volumes of fine tailings (hundreds of millions of cubic metres) cannot be easily incorporated into a dry landscape due to the fluid-like properties of the material. Currently, there is no single method of reclamation that could accommodate the volume of fine tailings produced and stored to date.

Two diverse methods are currently under consideration for reclaiming tailings into the landscape. The first method involves the incorporation of fine tailings into the bottoms of constructed lakes and is called the water-capping approach. The second method, called consolidated tailings, consists of adding gypsum to densified raw extraction tailings to which fine tailings have been added. Gypsum initiates the coagulation of the fine tailings fraction so that fine particles remain trapped within the coarse fraction from which fines-free water is released. Both methods would produce aquatic environments that would be influenced by oil sands related chemicals.

Elevated sulfate/chloride salinity, naphthenic acids, and polycyclic aromatic hydrocarbons are generally characteristic of oil sands tailing related water. Naphthenic acids are predominantly mono- and polycycloalkane carboxylic acids with aliphatic side chains of various lengths and have been identified as the family of compounds responsible for the acute lethality of fresh oil sands tailings water to fish (Verbeek 1994). Levels of priority, or parent polycyclic aromatic hydrocarbons, are generally relatively low as compared with the many alkane-substituted polycyclic aromatic hydrocarbons present. Experiments conducted using prototype ponds containing water-capped fine tailings at Syncrude's Mildred Lake site have demonstrated that primary production as well as macrophyte and invertebrate colonization proceed normally within 1–2 years of water capping (Boerger et al. 1992). Rainbow trout (*Oncorhynchus mykiss*) exposed to the overlying capping water within these ponds exhibited no acute toxicity over a period of several months (MacKinnon and Boerger 1986). However, chronic toxic effects, indirect effects such as limitations of primary and secondary productivity in the constructed systems, or nondirect effects such as the unsuitability of constructed habitats for particular life stages could compromise the viability of fish populations that will ultimately be exposed to oil sands tailings related waters. The potential of these factors to impact fish population status suggests the need for a diagnostic monitoring framework capable of determining which of the potential factors could be responsible for affecting fish populations.

Colby and Nepzy (1981) initiated such a framework based on the premise that fish population responses to particular stressors were distinct and could be used to predict fish population integrity. Munkittrick and Dixon (1989) and Gibbons and Munkittrick (1994) have refined the design to incorporate traditional fisheries data (growth, condition factor, age at maturity, reproductive potential, age and size structure) to compare potentially impacted with reference fish populations. Gibbons and Munkittrick (1994) have further classified fisheries data into three categories: age structure, energy expenditure, and energy storage.

The intent of this study was to examine the effects of the oil sands fine tailings water-capping option on survival and physiological indicators of energy storage and energy utilization in an indigenous species, yellow perch (*Perca flavescens*). A second goal of this study was to explore the utility of the monitoring framework based on physiological status measures by maintaining a single genetic stock and utilizing field stocking experiments. Finally, the results obtained were compared with data from natural yellow perch populations found within the same geophysical area as a means of examining the natural variability in measured parameters.

Materials and methods

Study site descriptions

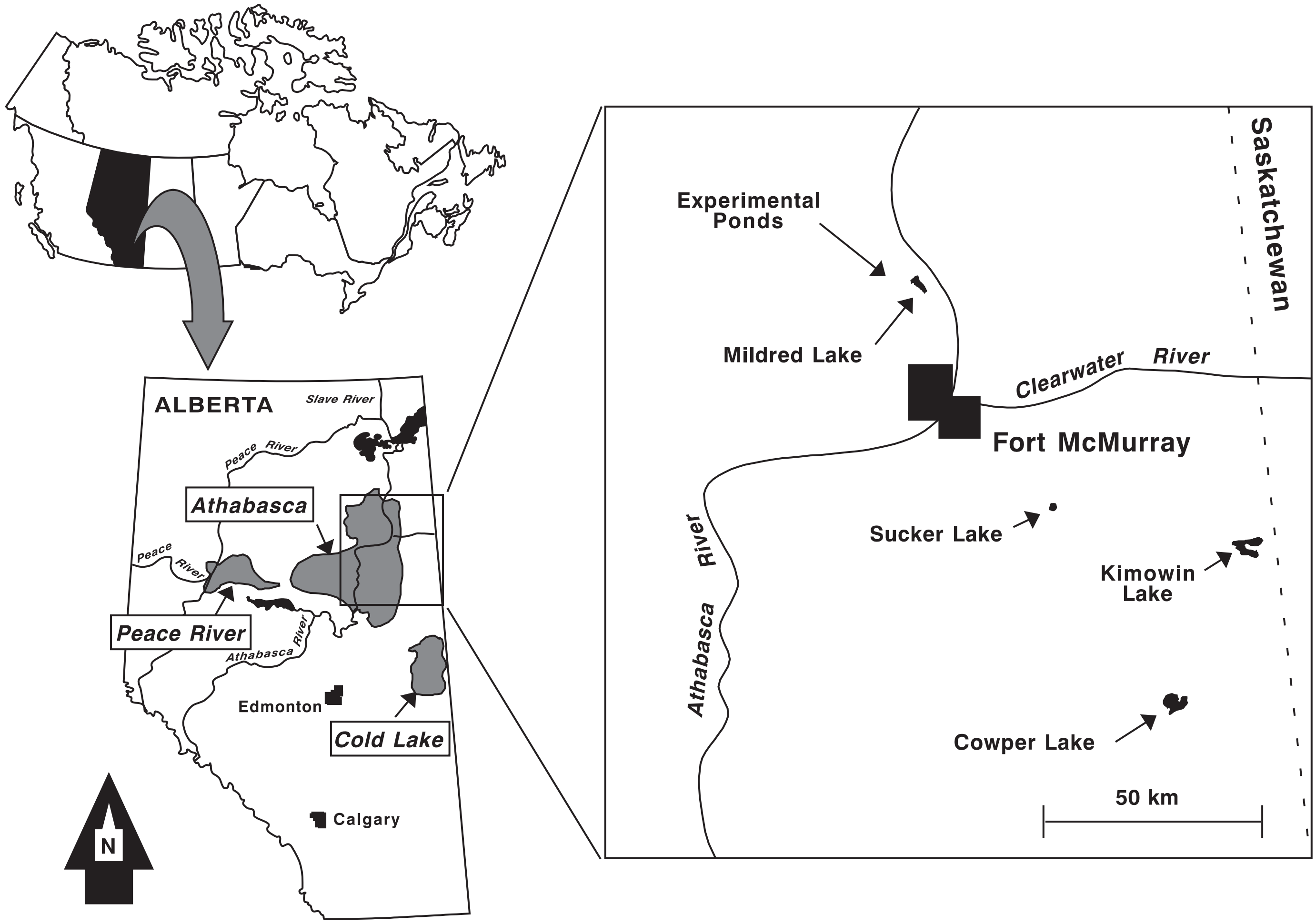
In 1993, a 4-ha pond was constructed in the experimental pond facility located on the northwest corner of the Syncrude Canada Ltd. oil sands lease area located on the Athabasca oil sands deposit north of Fort McMurray, Alberta (Fig. 1). About 70 000 m^3 of mature oil sands fine tailings were pumped from the main tailings storage pond into the fine tailings water-capping demonstration pond, hereafter called Demonstration Pond (Table 1). After the addition of tailings, 70 000 m^3 of natural surface water (drainage water from muskeg area) was pumped into Demonstration Pond from a nearby canal to a depth of 2.8 m above the tailings–water interface. A second pond, located immediately adjacent to Demonstration Pond, was constructed simultaneously. This pond was designed as a demonstration deep wetland, hereafter called Deep Wetland, and consisted of a dug-out pit filled with surface drainage water from the same source as Demonstration Pond. During the filling of these experimental ponds, fathead minnow (*Pimephales promelas*), brook stickleback (*Culaea inconstans*), and a small number of lake chub (*Couesius plumbeus*) were inadvertently pumped into the experimental ponds. All three species have survived to date in Demonstration Pond, whereas only fathead minnow remain in Deep Wetland.

A third experimental pond was formed in 1987 as a drainage basin in an area designated to be reclaimed as bison pasture on the south portion of the oil sands lease. The basin, and the area surrounding the pond, received overburden and materials deemed not suitable for reclamation. These materials were saline/sodic in nature and quite likely contained lean oil sands material. Reclamation (capping with muskeg) of the area surrounding this pond, hereafter called South Bison Pond, was completed in 1995. There were no fish species present in South Bison Pond at the initiation of this study. To provide a yellow perch forage base, 800 fathead minnow were transplanted from Deep Wetland to South Bison Pond in June 1995. Fathead minnow have subsequently formed a large and apparently viable population in this pond.

The lake chosen as the source of the experimental yellow perch was Mildred Lake, located on the Syncrude Canada Ltd. mining lease and present prior to the initiation of mining. Beginning in 1975, Mildred Lake was modified from its natural state and currently serves as a reservoir for plant makeup water that is pumped into Mildred Lake from the Athabasca River. Mildred Lake has been known to contain an abundant population of yellow perch since fisheries work was initiated in 1978.

To provide an ecologically relevant context for data derived from stocking experiments, the fish populations of three reference lakes, located off the oil sands lease or off the Athabasca oil sands

Fig. 1. Location of the Athabasca oil sands deposit and experimental sites within Alberta, Canada. Oil sands deposits are shaded.



deposit altogether, were evaluated. These natural lakes (Fig. 1; Table 1), Sucker Lake, Kimowin Lake, and Cowper Lake, were added at varying intervals throughout the course of the stocking experiments.

Experimental fish stocking

From June 26 to 28, 1995, 300 yellow perch were stocked into each of Demonstration Pond, Deep Wetland, and South Bison Pond. Yellow perch were captured in Mildred Lake using a Smith-Root 7.5-GPP gasoline-powered electrofisher mounted on a Smith-Root-18 electrofishing boat. Electrofisher voltage was set at 500 V pulsed direct current and pulse frequency was adjusted to provide a measured current between 2.4 and 2.8 A. Yellow perch captured by dip net were placed in a 500-L live well with flow-through water circulation. Upon capture of 75–100 yellow perch, fish were transferred to wire mesh holding cages (0.75 m^3) at a density of about 50 per cage. Yellow perch length and weight were recorded and several scales from just posterior to the pectoral fin were removed for ageing. Yellow perch were transported to the experimental ponds in two aerated circular polyethylene tanks with a capacity of about 0.75 m^3. Transportation time to the experimental ponds was less than 30 min (50–75 fish per tank).

The sex ratio of individuals taken from Mildred Lake in the spring of 1995 was highly skewed toward females (female to male sex ratio of 5:1). Fish sampled in the experimental ponds in the fall of 1995 reflected the stocking ratio. Apparent poor summer survival made obtaining sufficient numbers of yellow perch for the scheduled fall sampling difficult and required that yellow perch be restocked in 1996 to continue experimentation.

To correct for both survival and sex ratio problems, several modifications were made to fish capture and transfer procedures. The sex bias problem was corrected by capturing male yellow perch using overnight hoop net sets (May 20–22, 1996). Hoop nets were 1.22 m wide × 0.91 m high at the opening with a 7.6-m leader and 8-mm mesh size and no diagonal wings at the opening. Yellow perch can still be sexed externally at this time, since male yellow perch are still expressing milt. This capture technique used nearshore during the spawning period generally captured exclusively male yellow perch. A total of 250 male yellow perch were transferred to each of Demonstration Pond and South Bison Pond. Deep Wetland, which had become anoxic the previous winter, was not included in the 1996 experiments.

A further 500 yellow perch (presumed to be mostly female) were captured by electrofishing (as previously described) on June 7, 1996; 250 were transferred to each of Demonstration Pond and South Bison Pond. All yellow perch transfers in 1996 were made as rapidly as possible, with less than 1.5 h elapsing between removal from Mildred Lake and placement in the destination ponds. No data were recorded from the stocked fish in 1996 in order to minimize handling.

Water chemistry analysis

Temperature, pH, conductivity, and dissolved oxygen profiles were measured using a Hydrolab model 4041 water chemistry me-

Table 1. Experimental lake descriptions.

Site	Coordinates	Area (ha)	Maximum depth (m)	Known fish species	Sampling dates
Demonstration Pond	57°4.94′ N, 111°41.40′ W	4	2.9 (overlying 12 m of mature fine tailings)	YP, BS, FM, LC	F95, F96, W97
Deep Wetland	57°4.94′ N, 111°41.40′ W	4	6.0	YP, FM	F95
South Bison Pond	56°59.78′ N, 111°36.15′ W	5	5.0	YP, FM	F95, F96, W97
Mildred Lake	57°3.07′ N, 111°36.02′ W	170	8.2	YP, WE, LW, SPO, WS, LN, B	F95, F96, W97
Sucker Lake	56°25.34′ N, 110°51.73′ W	55	6.0	YP, NP, WS, SPO, BS, LC	F96, W97
Kimowin Lake	56°11.75′ N, 110°7.00′ W	448	12.0	YP, NP, BS	F96, W97
Cowper Lake	55°54.36′ N, 110°27.14′ W	600	5.5	YP, NP	W97

Note: Fish species: YP, yellow perch (*Perca flavescens*); FM, fathead minnow (*Pimephales promelas*); BS, brook stickleback (*Culaea inconstans*); LC, lake chub (*Couesius plumbeus*); NP, northern pike (*Esox lucius*); WE, walleye (*Stizostedion vitreum*); WS, white sucker (*Catostomus commersoni*); LN, longnose sucker (*Catostomus catostomus*); B, burbot (*Lota lota*); LW, lake whitefish (*Coregonus clupeaformis*); SPO, spottail shiner (*Notropis hudsonius*). Sampling dates: F95, fall 1995 (September 20–29, 1995); F96, fall 1996 (September 19–28, 1996); W97, winter 1997 (March 9–25, 1997).

ter. Temperature and dissolved oxygen were measured continually in Demonstration Pond using a turbidity probe for temperature (D&A Instruments, model OBS-3) and a low-demand oxygen probe (Greenspan model D1015) for dissolved oxygen.

The following chemical methods were used as outlined in American Public Health Association (1985) and U.S. Environmental Protection Agency (1979) standard method descriptions. Total nitrogen was measured using the Kjeldahl method. Total phosphate was analysed by ammonium molybdate – ascorbic acid – potassium tartrate colour reagent after acid–persulfate hydrolysis. Ammonia was quantified by means of the Bertelot reaction in an autoanalyser. Dissolved organic carbon was determined by measuring the carbon dioxide infrared absorbance of an ultraviolet light-oxidized solution against dissolved organic carbon standards. Carbonate levels were calculated from dissolved inorganic carbon concentrations. Dissolved inorganic carbon was determined by infrared measurement of carbon dioxide stripped from an acidified water solution. Major cations (sodium, potassium, calcium, magnesium) and trace metals were quantified by atomic emission using radiofrequency inductively coupled plasma. Samples for metals analyses were acidified without filtering.

The following chemical methods were used for the remaining water chemistry parameters. Major anions (chloride, sulfate) were analysed using ion chromatography followed by conductivity detection. The trace metals arsenic and selenium were analysed as hydrides using cold-vapour atomic absorption spectroscopy. Total naphthenates in water were quantified by FTIR spectroscopy following methylene chloride extraction of water.

Yellow perch survival estimates and sampling protocol

Quantitative estimates of yellow perch survival in the experimental ponds were not performed in 1995. During fall 1995 and spring 1996, qualitative estimates of yellow perch survival were observed while capturing stocked yellow perch for sacrifice. After stocking in 1996, catch-per-unit-effort (CPUE) was periodically measured in Demonstration Pond and South Bison Pond to quantify the relative density of yellow perch over 1 year. Four identical hoop nets spaced at random intervals were set overnight around the perimeter of the experimental ponds. Netting was carried out over two or three consecutive nights, changing net location each night to provide replicate estimates of relative density. Nets were always set with the leader anchored to shore and the net set perpendicular to shore. The number of fish per net and the exact fishing time were recorded in order to calculate CPUE as number of fish per net per hour. CPUE measurement was initiated 7 days after the completion of stocking in 1996 (June 21) and repeated during August and September 1996 as well as May 1997.

Yellow perch were sacrificed for physiological measurements from September 20 to 29 in 1995 and from September 18 to 28 in 1996. Yellow perch were captured in daytime or overnight hoop net or trap net sets (1.83 m with 19-mm mesh and a 46-m leader). To compare fall parameters with those from a period of more advanced reproductive development, yellow perch were also captured during late winter 1997 (March 9–27, 1997). Winter sampling was chosen because of the difficulties encountered while attempting to capture prespawning yellow perch in May 1996. At the majority of sites sampled during winter, multimesh, monofilament gill nets (50-m lengths) were set under the ice in daytime or overnight sets. Nets consisted of 10-m panels of 38.1-, 50.8-, 63.5-, 76.2-, and 88.9-mm stretch mesh. Gill nets were checked at least every 4 h. At Kimowin Lake and Cowper Lake, yellow perch were captured by angling through the ice.

Yellow perch were maintained alive until sampling and were dispatched with a sharp blow to the head. Length, weight, liver weight, gonad weight, gender, and sexual maturity were recorded. Operculae were placed in envelopes, kept cool, and later frozen pending age determinations. To age yellow perch, operculae were thawed, boiled to remove attached flesh, and dried prior to age determination. Anular rings were read under bright incandescent illumination. Ovary samples (about 1 g) for fecundity analysis were frozen in liquid nitrogen and later transferred to a –20°C freezer. In the laboratory, ovarian tissue was thawed and 50–200 mg was weighed out to 0.1 mg using a Sartorius digital balance. All ovarian follicles in the subsamples were counted. The weights of ovarian tissue used were chosen to correspond to between 150 and 300 follicles per sample. Total fecundity was calculated by multiplying the total count by the ratio of gonad weight to subsample weight.

Statistics

CPUE data were not normally distributed and were tested using

Table 1 (*concluded*).

Description
Contains 70 000 m^3 of mature fine tailings, capped with 70 000 m^3 of surface runoff water. Construction completed in October 1993
Dug-out area filled with surface runoff water. Completed in October 1993
Natural depression in reclaimed bison pasture
Natural lake on oil sands mining lease. Heavily modified in 1975 to act as a water reservoir. No angling pressure
Small boreal forest lake located in the Athabasca River basin. May receive drainage water from lakes overlying the oil sands deposit. No angling pressure
Medium-sized boreal forest lake located in the Beaver River basin 40 km west of the Athabasca oil sands deposit. Heavy angling exploitation in winter only
Medium-sized boreal forest lake located in the Athabasca River basin 30 km south (upstream) of the Athabasca oil sands deposit. Heavy angling exploitation in winter only

the nonparametric Mann–Whitney *U* test. Condition factor, liver size, gonad size, and fecundity data were analysed using analysis of covariance (ANCOVA) on base-10 logarithmically transformed variables. Corrected total weight (body weight – liver weight – gonad weight) was used for all statistical comparisons and calculations. It should be noted that although statistical comparisons using ANCOVA were completed on body, liver, and gonad weight and also total fecundity, data are presented as somatic indices to facilitate comparisons between sampling periods and between studies. Fulton's condition factor was calculated as body weight/length^3 × 100, gonadosomatic index as gonad weight/body weight × 100, liver somatic index as liver weight/body weight × 100, and fecundity index as total number of eggs/body weight.

All statistical analyses of physiological indices were restricted to comparing Mildred Lake yellow perch with the yellow perch from the stocked ponds. Only yellow perch with actively maturing gonads were included in the analyses of the indices. Statistical analyses were performed this way due to multiple interactions (heterogeneity of slope) when off-site references were included, to keep statistical power similar at all sampling time points, and to simplify statistics in order to concentrate on the a priori hypotheses of the stocking experiment. Qualitative comparisons were deemed sufficient to assess the relevance of measured indices with regard to reference populations. All statistical testing was completed using the SYSTAT software package (Wilkinson 1990). The critical level of statistical differences for all analyses in this paper was assessed at $\alpha = 0.05$.

Results

Water chemistry

Water chemistry characteristics indicated a gradient of both organic compounds and major ions as shown by the measured water quality parameters (Table 2). The higher salinity in oil sands associated ponds was characterized by sulfate as the major anion followed by chloride. Sodium was the major cation, but elevations in potassium, calcium, and magnesium were also evident. The only indicator of elevated oil sands related organic compounds shown was total naphthenates. Total naphthenates demonstrated the same overall gradient as major ions. South Bison Pond water chemistry was influenced most, despite the lack of any oil sands tailings in the area, followed by Demonstration Pond and Deep Wetland. Oil sands also showed a gradient of influence in the reference lakes in this study. Mildred Lake had low levels of naphthenates as well as slightly elevated levels of chloride and sulfate. Sucker Lake had no apparent major ion effects but showed traces of naphthenates. Kimowin Lake, the only site entirely removed from the Athabasca oil sands deposit for which we had extensive water chemistry data, did not have detectable concentrations of naphthenates.

Oil sands exposed sites were also characterized by some elevations in levels of trace metals in water (Table 3). Increases in boron, lithium, and strontium followed the same gradient as salinity and naphthenate levels, indicating a probable association with oil sands and related clays. Zinc also showed a slight increase, particularly in Demonstration Pond. Levels of lead and selenium (not shown in Table 3) were also measured and found to be nondetectable in all samples analysed (<20 and <0.2 $\mu g \cdot L^{-1}$, respectively).

Temperature profiles of the experimental ponds and reference sites (data not shown) showed a nearly identical regime among the lakes and ponds. Dissolved oxygen profiles were more variable, with some sites demonstrating late winter oxygen deficits. Deep Wetland in particular had a significant fish kill during winter 1995–1996. All yellow perch and greater than 99% of fathead minnow perished. Sucker Lake had oxygen levels of 2.0 $mg \cdot L^{-1}$ in March 1997; however, a substantial number of yellow perch survived this event. All other ponds showed levels of late winter dissolved oxygen in excess of 60% saturation. Mildred Lake, which receives aeration through a continuous input of water pumped up from the Athabasca River, had greater than 85% saturation.

Perch survival

Yellow perch capture effort in the experimental ponds in fall 1995 indicated that Deep Wetland yellow perch had the best overall summer survival. Capture attempts in Demonstration Pond and South Bison Pond provided barely enough yellow perch for the sampling needs (nominally 20 males and 20 females), whereas Deep Wetland yielded greater than 150 yellow perch in one night of netting (representing 50% of the total number stocked). In 1996, CPUE (Fig. 2) was utilized to quantify survival. Demonstration Pond showed a significant reduction in CPUE between 1 week poststocking and August, after which the CPUE data were relatively stable. South Bison Pond also showed an initial decrease, but statistically significant differences did not occur until May 1997 (after about 80 yellow perch had been sacrificed for sampling purposes during fall 1996 and winter 1997). The yellow perch stocked into Demonstration Pond and South Bison Pond demonstrated superior survival in 1996 as opposed to the 1995 stocking effort.

Perch condition indices

Yellow perch condition factor responses to being stocked in the experimental ponds were sex dependent. Male condition factor remained unaffected by stocking, showing no statistically significant differences from the condition factor

Table 2. Mean water chemistry parameters ($mg \cdot L^{-1}$) (*n*, SEM) for experimental ponds and lakes during the open-water season of 1995 and 1996.

Site	pH	Cond. ($\mu S \cdot cm^{-1}$)	TDS	Na^+	K^+	Mg^{2+}	Ca^{2+}	Cl^-
1995								
Deep Wetland	8.7	489	348	45.3				6.9
	(3, 0.1)	(2, 19)	(2, 17)	(5, 1.5)				(3, 0.1)
Demonstration Pond	8.4	826	557	126	4.8	16.0	48.1	31.2
	(5, 0.2)	(5, 10)	(4, 4)	(6, 3.7)	(4, 0.4)	(4, 0.2)	(4, 0.7)	(6, 0.6)
South Bison Pond	7.9	3131	2392	430	12.5	87.9	200	116
	(1)	(1)	(1)	(1)	(1)	(1)	(1)	(1)
1996								
Deep Wetland	8.4	421	297	44.6	3.1	18.9	34.8	6.2
	(5, 0.1)	(5,9)	(5, 8)	(5, 1.2)	(5, 0.22)	(5, 0.6)	(5, 3.1)	(5, 0.2)
Demonstration Pond	8.5	788	565	139	4.3	16.0	45.9	36.1
	(9, 0.1)	(9, 7)	(9, 5)	(9, 2.8)	(9, 0.38)	(9, 0.3)	(9, 2.3)	(9, 0.9)
Mildred Lake	7.7	288	194	18.1	1.7	9.1	39.3	7.2
	(3, 0.3)	(3, 14)	(3, 7)	(3, 2.0)	(3, 0.1)	(3, 0.4)	(3, 2.2)	(3, 1.0)
South Bison Pond	8.0	1825	1594	246	8.4	57.1	132	78.6
	(5, 0.1)	(5, 94)	(3, 12)	(5, 15.6)	(5, 0.62)	(5, 4.1)	(5, 9.2)	(5, 3.6)
Sucker Lake	7.3	176	134	8.5	1.4	6.6	23.6	0.6
	(3, 0.1)	(3,)	(2, 16)	(2, 0.1)	(1)	(1)	(1)	(3, 0.3)
Kimowin Lake	7.7	242	171	11.7	2.8	11.7	31.2	0.4
	(3, 0.2)	(2, 1)	(2, 5)	(3, 0.1)	(3, 0.3)	(3, 0.4)	(3, 2.3)	(3, 0.1)

Note: Empty cells indicate that the parameter was not measured during the indicated period. TDS, total dissolved solids; TN, total nitrogen; TP, total phosphorus; DOC, dissolved organic carbon.

Table 3. Mean water trace metal concentrations ($\mu g \cdot L^{-1}$) (*n*, SEM) for experimental ponds and lakes during the open-water season of 1995 and 1996.

Site	Al	Sb	As	Ba	Be	B	Cd	Cr	Co	Cu	Fe
1995											
Demonstration Pond			1.0			415				1.5	
			(1)			(3, 72)				(3, 0.5)	
South Bison Pond			0.8	50			<2	<2			30
			(1)	(1)							(1)
1996											
Deep Wetland	20	<0.2	1.0	23	<1	80	<3	5	4.0	2.0	20
	(4, 5)		(2, 0.2)	(4, 3)		(4, 20)		(4, 3)	(4, 1)	(4, 1)	(4, 10)
Demonstration Pond	72	<0.2	1.0	40	2	484	<3	3	3.6	2.2	62
	(5, 22)		(2, 0.2)	(5, 4)	(5,1)	(5, 12)		(5, 1)	(5, 0.5)	(5, 0.4)	(5, 40)
Mildred Lake	<10	<0.2	0.4	67	3	37	<3	<2	<3	<1	143
			(1)	(3, 2)	(3, 1)	(3, 3)					(3, 50)
South Bison Pond	55	0.3	<0.3	40	3	637	<3	<2	3.8	2.5	18
	(4, 33)	(2, 0.1)		(4, 4)	(4, 1)	(4, 22)			(4, 0.6)	(4, 0.6)	(4, 7)
Sucker Lake	20			20	<1	40	5	6	<3	3	400
	(1)			(1)		(1)	(1)	(1)		(1)	(1)
Kimowin Lake	15			27	3	43	<3	3	<3	<1	20
	(3, 10)			(3, 3)	(3, 1)	(3, 5)		(3, 1)			(3, 5)

Note: Empty cells indicate that the parameter was not measured during the indicated period.

measured in Mildred Lake in each of the sample periods (Table 4). Female condition factor was higher, and statistically different, in yellow perch from the experimental ponds when compared with Mildred Lake in the falls of 1995 and 1996. Winter 1997 samples indicated no statistically significant differences among the three sites. Condition factors for both sexes in Mildred Lake and the experimental ponds fell within the range of variability shown in yellow perch captured in the off-site reference lakes. The range is characterized by the values computed for the stunted population of Sucker Lake (<1.2) and the population of Kimowin Lake (>1.5).

Liver size of stocked fish was consistently larger than that of Mildred Lake fish, the exception being that liver size in Deep Wetland males was unaffected in fall 1995 (Table 4). Yellow perch liver sizes in South Bison Pond and Demon-

Table 2 (*concluded*).

SO_4^{2-}	CO_3^{2-}	NH_3^{+}	TN	TP	Total naphthenates	DOC
43.6	286	0.10	1.64	0.07	3.0	34.6
(3, 3.1)	(3, 10)	(3, 0)	(3, 0.02)	(3, 0.02)	(1)	(3, 1.6)
98.8	422	0.09	2.49	0.06	8.5	38.2
(6, 2.9)	(6, 12)	(6, 0.01)	(5, 0.61)	(5, 0.01)	(2, 1.1)	(6, 1.0)
1160	386					
(1)	(1)					
39.6	208	0.04	1.26	0.05	1.4	28.9
(5, 0.7)	(5, 11)	(5, 0.01)	(5, 0.06)	(5, 0.01)	(3, 0.2)	(5, 2.0)
97.9	335	0.07	2.48	0.07	5.6	35.8
(9, 0.9)	(6, 9)	(9, 0.01)	(7, 0.91)	(7, 0.02)	(5, 0.6)	(9, 1.3)
21.8	134	0.03	0.80	0.06	1.1	15.2
(3, 2.2)	(3, 4)	(3, 0.01)	(3, 0.1)	(3, 0.01)	(2, 0.1)	(3, 1.2)
747	184	0.06	0.91	0.05	9.1	30.7
(5, 38.4)	(5, 6)	(5, 0.02)	(5, 0.08)	(5, 0.01)	(3, 2.4)	(5, 1.9)
4.6	118	0.11	3.25	0.02	0.4	24.5
(3, 0.4)	(1)	(2, 0.07)	(1)	(1)	(1)	(3, 1.6)
3.6	175	0.06	3.68	0.06	<0.3	17.2
(3, 0.8)	(3, 4)	(3, 0.02)	(2, 1.64)	(2, 0.03)	(1)	(3, 1.2)

Table 3 (*concluded*).

Li	Mn	Mo	Ni	Sr	Ti	V	Zn
43				340			9
(1)				(1)			(1)
			17				
11	20	<3	6	210	5	<2	10
(4, 1)	(4, 3)		(4, 1)	(4, 10)	(4, 2)		(4, 5)
36	4	6	7	332	4	<2	24
(5, 1)	(5, 1)	(5, 2)	(5, 1)	(5, 7)	(5, 1)		(4, 9)
6	56	<3	11	230	5	<2	4
(3, 1)	(3, 15)		(3, 3)	(3, 9)	(3, 1)		(3, 1)
79	19	<3	<5	1323	4	3	12
(4, 4)	(4, 7)			(3, 52)	(4, 1)	(4, 1)	(4, 5)
8	168	<3	<5	96	5	5	1
(1)	(1)			(1)	(1)	(1)	(1)
11	38	<3	<5	138	<3	<2	5
(3, 1)	(3, 14)			(3, 5)			(3, 2)

stration Pond were significantly larger in both sexes in winter 1997 when compared with either Mildred Lake fish or the upper extreme of the off-site reference lakes.

Reproductive indices and observations

Gonad size changes were also apparent in yellow perch stocked in the experimental ponds as compared with Mildred Lake yellow perch (Fig. 3). In fall 1995, male and female yellow perch from Deep Wetland had significantly larger gonad size than Mildred Lake yellow perch. Male and female yellow perch from the other experimental ponds showed no significant differences. In fall 1996, male and female yellow perch from Demonstration Pond and South Bison Pond both had significantly larger gonad size than Mildred Lake yellow perch. In female yellow perch, condition factor differences were reduced and gonad size differ-

Table 4. Mean (SEM) age, length, weight, condition factor, liver somatic index (LSI), percent spawning periodicity, and sample size for yellow perch used in this study.

	Females						
Capture location	Age (years)	Length (cm)	Weight (g)	Condition factor	LSI	Spawning periodicity (%)	*n*
Fall 1995							
Demonstration Pond	7.2	22.7	164	1.30	2.08	0	19
	(0.6)	(0.4)	(9)	(0.02) *a*	(0.05) *a*		
Deep Wetland	7.7	22.7	171	1.29	1.63	0	20
	(0.5)	(0.6)	(14)	(0.02) *a*	(0.07) *b*		
South Bison Pond	8.0	22.2	14(7)	1.27	1.88	0	21
	(0.6)	(0.3)		(0.02) *a*	(0.08) *ab*		
Mildred Lake	8.2	28.7	14(6)	1.19	1.30	19.0	47
	(0.3)	(0.3)		(0.01) *b*	(0.05) *c*		
Fall 1996							
Demonstration Pond	8.5	22.9	15	1.21	1.76	0	19
	(0.6)	(0.3)	(6)	(0.02) *ab*	(0.09) *a*		
South Bison Pond	8.7	23.8	185	1.27	1.68	0	20
	(0.6)	(0.4)	(10)	(0.02) *a*	(0.06) *a*		
Mildred Lake	7.9	22.9	148	1.16	1.31	24.1	23
	(0.5)	(0.4)	(7)	(0.02) *b*	(0.05) *b*		
Sucker Lake	4.5	18.4	79	1.18	1.39	0	27
	(0.2)	(0.4)	(6)	(0.01)	(0.07)		
Kimowin Lake	8.3	28.7	422	1.64	1.56	0	39
	(0.5)	(0.4)	(16)	(0.02)	(0.03)		
Winter 1997							
Demonstration Pond	6.7	22.0	165	1.25	3.04	0	20
	(0.4)	(0.4)	(9)	(0.02) *a*	(0.12) *a*		
South Bison Pond	8.1	24.5	230	1.23	2.96	0	21
	(2.0)	(0.4)	(11)	(0.01) *a*	(0.06) *a*		
Mildred Lake	7.7	21.8	147	1.21	1.67	12.1	29
	(0.3)	(0.3)	(6)	(0.02) *a*	(0.04) *b*		
Sucker Lake	4.0	18.8	93	1.15	2.20	9.1	20
	(0.1)	(0.4)	(5)	(0.01)	(0.08)		
Kimowin Lake	9.4	28.3	440	1.51	2.64	0	20
	(0.8)	(0.6)	(28)	(0.03)	(0.10)		
Cowper Lake	6.7	28.0	363	1.38	2.11	25.0	18
	(0.3)	(0.3)	(13)	(0.02)	(0.09)		

Note: Ages reflect the exact ages of the fish at the beginning of the previous growing season. Means within a column for a particular sampling date followed by the same letter are not significantly different ($p > 0.05$).

ences were accentuated over the period of time between the fall 1996 and the winter 1997 sample collections. Gonad sizes in Mildred Lake and the experimental ponds fell within, or exceeded, the range found in the off-site reference locations, with Sucker Lake once again being on the low end of the distribution and Kimowin Lake on the high end.

Fecundity estimates (Fig. 4) generally parallelled the gonad size data, indicating that during these sampling periods, increases in gonad size were due to an increase in the number of ovarian follicles and not a change in the follicle size. Only one example of the opposite possibility was observed in fall 1995 for South Bison Pond. Gonad size in South Bison Pond yellow perch was significantly lower than that measured in Deep Wetland and Demonstration Pond, but fecundity remained unaffected as a result of a decrease in follicle size. Fecundity data of yellow perch from the experimental ponds generally fell within, or exceeded, the fecundity estimates observed at the off-site reference sites.

During the spring of 1997, yellow perch stocked in the experimental ponds in 1996 were observed to undergo spawning in a normal fashion as indicated by the deposition of egg strings on small spruce trees introduced as spawning substrate. CPUE netting revealed that a large majority of the female yellow perch had spawned by the completion of these netting efforts in late May. The timing of spawning corresponded well to what was observed in Mildred Lake during spring 1997.

Of particular note was the appearance of spawning periodicity at some of the sites (Table 4). Spawning periodicity, or "resting", can be defined as the absence of any gonadal development in an individual capable of such development. In this case, female yellow perch that would not be spawning at age 4 or greater were evaluated as having a resting year. Since female gonads remain arrested in their previtellogenic state, spawning periodicity is easily observed by early September. Mildred Lake showed this phenomenon over the pe-

Table 4 (*concluded*).

Males					
Age (years)	Length (cm)	Weight (g)	Condition factor	LSI	*n*
7.1	19.9	108	1.24	1.44	7
(1.0)	(0.5)	(10)	(0.03) *a*	(0.10) *ab*	
7.4	19.2	99	1.29	1.26	13
(0.7)	(0.1)	(3)	(0.02) *a*	(0.06) *bc*	
7.3	19.3	102	1.29	1.63	6
(1.1)	(0.6)	(11)	(0.03) *a*	(0.11) *a*	
7.1	19.2	96	1.26	1.10	14
(0.5)	(0.3)	(5)	(0.02) *a*	(0.05) *c*	
7.1	18.5	86	1.25	1.71	19
(0.3)	(0.2)	(3)	(0.01) *a*	(0.05) *a*	
7.6	19.6	104	1.27	1.25	21
(0.3)	(0.2)	(4)	(0.01) *a*	(0.10) *b*	
7.0	18.7	87	1.23	1.14	20
(0.6)	(0.2)	(4)	(0.01) *a*	(0.03) *c*	
4.0	15.9	53	1.17	1.41	20
(0.2)	(0.4)	(0.4)	(0.01)	(0.03)	
10.8	26.4	324	1.58	1.14	17
(0.7)	(0.8)	(22)	(0.04)	(0.07)	
5.9	18.1	81	1.24	3.18	21
(0.4)	(0.3)	(4)	(0.02) *a*	(0.21) *a*	
7.8	19.8	109	1.25	2.79	13
(0.6)	(0.3)	(7)	(0.02) *a*	(0.10) *a*	
8.2	17.7	78	1.27	1.81	12
(1.0)	(0.6)	(7)	(0.02) *a*	(0.13) *b*	
4.0	17.0	61	1.14	1.65	20
(0.1)	(0.4)	(4)	(0.01)	(0.09)	
8.2	23.4	248	1.50	2.13	22
(1.0)	(1.2)	(29)	(0.04)	(0.11)	
4.9	20.8	163	1.33	1.87	18
(0.6)	(1.3)	(31)	(0.03)	(0.11)	

riod of years framed by this study with frequencies ranging from 12 to 24% of the female population. Female yellow perch transferred from Mildred Lake to the experimental ponds that were age 4 and above did not show a single case of missing a reproductive season (n = 140 females examined), nor did any of the Kimowin Lake females examined (n = 59). However, periodicity was observed in female yellow perch at Sucker Lake and Cowper Lake.

Summary of physiological indices

Physiological indicator data have been aggregated and summarized in Table 5 as energy storage parameters (condition factor, liver size) and energy utilization parameters (gonad size, fecundity, spawning periodicity). Changes in the parameters relative to the lake of origin for the stocked yellow perch (Mildred Lake) over the study period provided a potentially useful means of assessing the impacts of experimental pond water exposure on stocked yellow perch. Changes are indicated only if all component measures of the parameter showed a significant change in the same direction. Response patterns between sites, for different stressor regimes, varied and were not consistent in maintaining unique response patterns. In no cases did the energy storage or utilization parameters show a significant negative impact (i.e., parameter decrease).

Discussion

Experiments on yellow perch stocked in ponds containing oil sands related reclamation water revealed no negative changes in somatic indices as a result of exposure. Yellow perch showed equal or superior ability to store and utilize energy for reproductive tissue growth relative to the lake from which they originated. Somatic indices in yellow perch from exposed areas generally fell within the range of indices measured at off-site reference lakes. Typically, the magni-

Fig. 2. Mean CPUE estimates performed using hoop nets in Demonstration Pond and South Bison Pond following the 1996 stocking effort. Error bars represent standard error of the mean. Asterisks indicate statistically significant differences between means ($p < 0.05$). The number at the base of each bar indicates the quantity of replicate net sets.

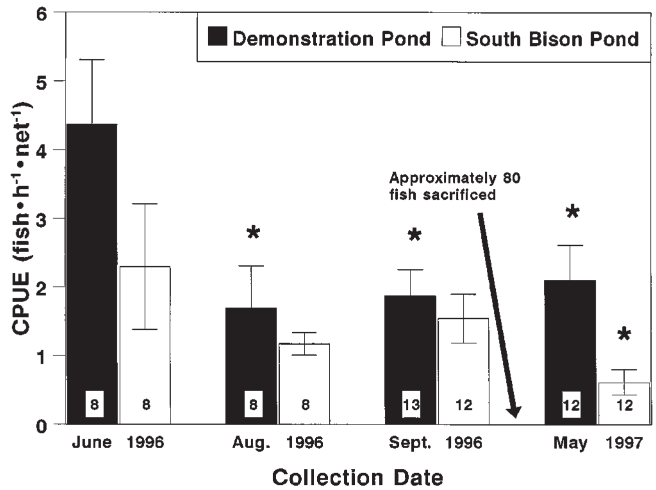
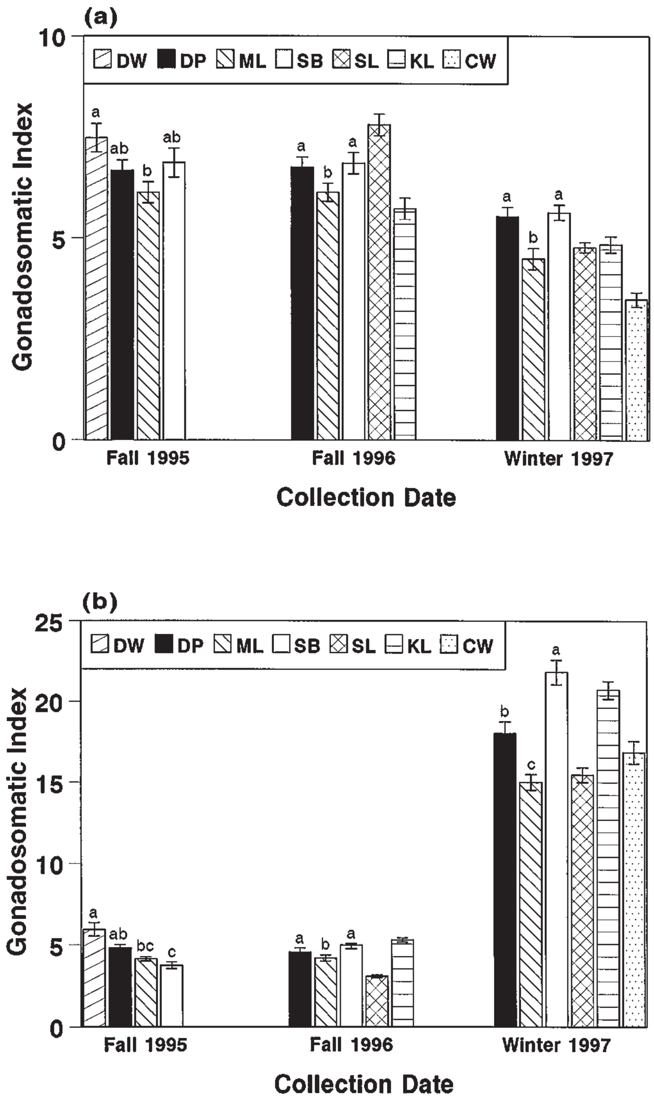


Fig. 3. Mean gonadosomatic index in (*a*) male and (*b*) female yellow perch captured at the experimental locations during fall 1995, fall 1996, and winter 1997. Error bars represent standard error of the mean. Means within sampling periods with the same letter are not significantly different ($p > 0.05$). Lake abbreviations as per Table 1.



tude of differences between reference lake measurements exceeded changes that occurred in yellow perch stocked in experimental ponds.

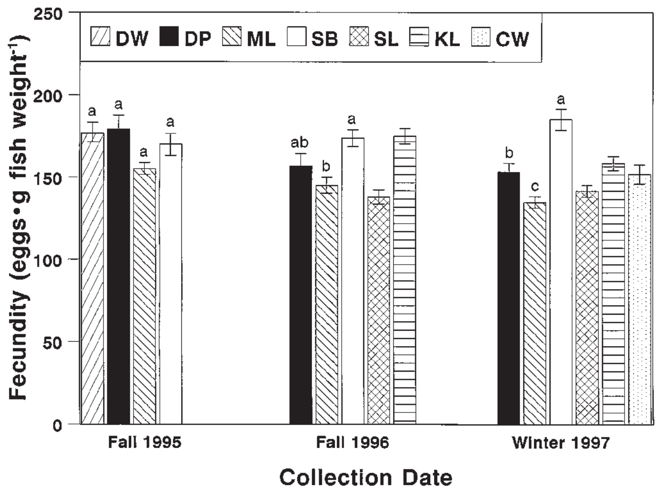
Several chemical indicators of oil sands impacts on water quality were observed in this study. Increased salinity, with sulfate being the major anion, appears to be common in oil sands from this particular source. Sulfate salinity was closely parallelled by levels of naphthenic acids and by the trace metals boron, lithium, and strontium. In the sites tested, none of these marker compounds appeared to be present at sufficiently high levels to cause overt levels of acute lethality in adult yellow perch. However, levels of salinity in South Bison Pond were within a factor of 2 of the estimated 2000–4000 mg total dissolved solids$\cdot L^{-1}$ (sulfate saline) that yellow perch embryos can tolerate (Koel and Peterka 1995).

Initial declines in the numbers of yellow perch present in the experimental ponds were likely attributable to a delayed response precipitated by the trauma of capturing and transporting the fish. To what level this stress was influenced by the water chemistry of the experimental ponds remains unknown. Observed increases in 1996 poststocking survival probably resulted from efforts made to minimize handling and captivity time of yellow perch during stocking. The duration of the stocking experiments to date does not allow comment on possible long-term effects; further studies are required before conclusions about long-term risks can be drawn.

In this study, yellow perch, which are abundant in many boreal lakes and have a large geographic distribution and a home range limited to less than 1 km^2 (Fish and Savitz 1983), proved suitable for use within the context of the environmental framework developed by Munkittrick and Dixon (1989) and Gibbons and Munkittrick (1994). Given the short duration of the experiments reported here (2 years) relative to the life span of yellow perch (10+ years), it was not possible to assess changes in age distribution resulting from exposure to the experimental systems. The requirement for such data represents a practical limitation on the utility of monitoring frameworks for assessing the possible impacts of pulse events (e.g., tailings pond spills or flood events) and limits their use to situations where known stresses have been acting in a consistent manner for a period at least as long as a single, complete life cycle for the studied species.

A further, and quite serious, limitation of monitoring frameworks based on fisheries parameters is the choice of reference locations. For example, the recommendations of Canada's environmental effects monitoring protocol for fish require that one reference site be chosen as a basis for comparison with a possibly impacted site. The data from the off-site reference lakes presented in this study show that the use of a single reference location is at best misguided. In this study, the fortuitous choice of two off-site reference lo-

Fig. 4. Mean fecundity index in female yellow perch captured at the experimental locations during fall 1995, fall 1996, and winter 1997. Fecundity is expressed as number of eggs per gram of organ-corrected fish weight. Error bars represent standard error of the mean. Means within sampling periods with the same letter are not significantly different ($p > 0.05$). Lake abbreviations as per Table 1.



cations that reflect large differences in the measured fisheries parameters clearly demonstrates this problem. If the yellow perch monitoring parameters from either Sucker Lake or Kimowin Lake were compared directly with Demonstration Pond yellow perch parameters, two opposite conclusions would result. These results strongly indicate the need to know and understand the natural range of variability in fisheries parameters. Unfortunately, in order to accomplish this, a large number of sites need to be surveyed; this is both expensive and time consuming.

Despite the limitations of the monitoring strategy examined here, energy storage and utilization parameters were successfully used to characterize the response of stocked yellow perch to the experimental systems. Energy storage indices (condition factor, liver weight) showed no change or increased in yellow perch stocked in the experimental systems. The results suggest improved resource availability and a lack of any observable problems with adult yellow perch gonadal growth in the stocked ponds as compared with Mildred Lake. Examination of available diet, density, and competition data provides direct support for this conclusion. Yellow perch from the experimental ponds had the highest proportion of fish in their diet (Gould 1999) and were exceeded in condition only by fish from Kimowin Lake, whose diet consisted almost exclusively of abundant and calorie-rich prey (*Gammarus pulex* and *Hyalella azteca*). Sucker Lake yellow perch had the highest densities of all studied populations (authors' unpublished data). The population was dominated by large numbers of small fish, a pattern suggestive of food limitation and high intraspecific competition. The presence of white sucker (*Catostomus commersoni*), a known interspecific competitor, further suggests that Sucker Lake yellow perch were resource limited (Johnson 1977; Post and Cucin 1984).

The magnitude of condition factor differences in female yellow perch was reduced between fall 1996 and winter

Table 5. Relative changes in energy storage and utilization parameters (in that order) for yellow perch from the three experimental ponds as compared with Mildred Lake, the lake of origin for the stocked yellow perch.

	Deep Wetland	Demonstration Pond	South Bison Pond
Females			
Fall 1995	+ +	+ 0	+ 0
Fall 1996		0 +	+ +
Winter 1997		0 +	0 +
Males			
Fall 1995	0 +	+ 0	0 0
Fall 1996		0 +	0 +
Winter 1997		0 +	0 +

Note: +, significant increase; 0, no change; –, significant decrease. Changes are indicated only if all indices show a significant change in the same direction.

1997 and any statistical differences disappeared as a result. This suggested a seasonal change in condition factor. Such a change would be expected to occur if stored somatic energy reserves were utilized for gonad growth, since 75% of gonad growth occurs between fall and late winter. Kimowin Lake also showed a substantial drop in condition factor over this period. In contrast with changes observed in condition factor, liver size in yellow perch from the experimental ponds increased substantially between fall 1996 and winter 1997. Experimental pond yellow perch liver size was also larger than that of yellow perch in the off-site reference locations, particularly during winter 1997. The liver serves as the primary site of chemical metabolism and excretion and also as an energy storage organ. There is evidence to suggest that liver size may increase due to a petroleum-related chemical challenge under conditions where energy intake is controlled (Steadman et al. 1991; Truscott et al. 1992). Increased mixed-function oxygenase enzyme activity and bile polycyclic aromatic hydrocarbon metabolites in experimental pond yellow perch would suggest that this may be the case (van den Heuvel et al. 1999).

Energy utilization indices, gonad size, and fecundity for experimental pond yellow perch either showed significant increases or, when no significant change occurred, were higher than similar values derived for Mildred Lake yellow perch. The exception to this was South Bison Pond yellow perch in fall 1996. South Bison Pond yellow perch fecundity was similar to Mildred Lake yellow perch fecundity during this same period, indicating that smaller ovarian follicles relative to the other sites were responsible for the reduced gonad size. In 1996, yellow perch in South Bison Pond reversed this trend and had the highest fecundity and gonad size of any site examined. The reasons for this anomaly are unknown; however, fathead minnow were stocked during the same period of 1995 as the yellow perch. A rapid increase in the size of the fathead minnow population occurred between 1995 and 1996, possibly resulting in a better food base for yellow perch. It is also possible that the improved food source masked toxicological effects on gonad growth in 1996 but not in 1995.

Differences in gonad size in females were accentuated in winter 1997, as opposed to fall 1996, just as condition factor differences disappeared over this period. This suggests that yellow perch in the experimental pond were successful at converting energy stored in the body into reproductive tissue. Thus, exposure to oil sands related reclamation waters does not appear to cause any deficiency in the metabolic redistribution of energy, as has been observed in white sucker exposed to pulp mill effluent (Munkittrick et al. 1994).

An unexpected phenomenon was the presence of resting years in reproduction or spawning periodicity in yellow perch from several of the locations examined. Periodicity could be considered a major energy utilization problem or merely the failure to accumulate sufficient energy to cue the initiation of gonadal recrudescence in the fall. Yellow perch stocked in the experimental ponds did not show spawning periodicity and appeared to have better energy stores than yellow perch from Mildred Lake. This would seem to suggest that the lack of gonad development in the resting fish was indeed related to food supply. However, two observations argue against this hypothesis: Sucker Lake yellow perch probably had the highest density and poorest food availability, yet periodicity was only infrequently observed. In addition, Cowper Lake yellow perch grew to sizes similar to Kimowin Lake yellow perch and showed the second highest condition factors in winter 1997 female yellow perch, yet showed a similar rate of spawning periodicity to that of Mildred Lake yellow perch. The observation of resting years is observed in other fish species, particularly at higher latitudes, and also shows contradictory data as to possible causes (Pulliainen and Korhonen 1993).

The results of this study indicate that the exposure of yellow perch to waters associated with oil sands reclamation does not compromise the short-term physiological status of exposed individuals. Irrespective of improved food supply in the experimental ponds, there was no direct evidence to suggest that oil sands related compounds were responsible for reduced survival, condition, or gonadal growth in yellow perch. It should be noted that physiological indices in adult fish may not be the most sensitive indicators of toxicant stress. Impacts may be masked by compensatory responses at both the individual and population level. This would imply that monitoring frameworks based on adult fish may not be the best tool for predicting potential risks to a population. Early life stages are generally thought to be a more sensitive stage of development and are therefore more likely to impact upon population status than are small changes in somatic indices at the adult stage of development.

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