Restoration of a Mine Pit Lake from Aquacultural Nutrient Enrichment

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Abstract

Minnesota, the land of 10,000 lakes, also has more than 4000 abandoned quarry pits and over 200 deep, exhausted iron ore pits. In the past 25 years the iron ore pits have gradually filled with groundwater and surface water, forming lakes on the Cuyuna, Mesabi, and Vermillion Iron Ranges in northeastern Minnesota. Most remain abandoned, but besides creating a small number of recreational parks and fisheries, the regional economic development agency promoted approximately 20 of the pit lakes for economic reclamation by using them for salmonid aquaculture. Intensive net-pen aquaculture was carried out from 1988 to 1995 in the Twin City–South and Sherman pit lakes on the Mesabi Range. A water quality controversy resulted over the potential for long-term degradation of the lakes and regional aquifer. The Minnesota Pollution Control Agency then mandated that aquaculture be terminated in Twin City–South in May 1993 and the lake restored to preaquaculture conditions by 1996. With no management other than artificial aeration for one summer, the lake rapidly recovered to near baseline water quality and returned to an oligo-mesotrophic (unproductive) status. Within 18 months the phosphorus budget was typical of reference pit lakes in the area and dissolved oxygen in bottom water remained above ~4 mg O_2/L without artificial aeration. Algal growth was low in 1993, due to light limitation from artificial mixing, but it remained low in 1994 without any management due to renewed phosphorus limitation. Inorganic nitrogen initially decreased faster than expected, at a rate similar to its increase during intensive aquaculture. More rapid reductions in water column nutrients might have occurred in 1993 by reducing aeration to allow anoxia in the lower hypolimnion, promoting denitrification and minimizing sediment resuspension, but this was precluded by water quality standards. The "natural" burial of solid wastes under inorganic sediment eroded from the basin walls effectively minimized transport of sediment nutrients to the overlying water. Fallowing for several years provided a simple, effective method for restoration of these pit lakes from aquacultural impacts. No change attributable to aquaculture was observed in the water quality of three nearby pit lakes, including a drinking water source. This fact suggests that there were few or no impacts from off-site migration of aquaculturally enriched water into the regional aquifer.

Introduction

The landscapes of industrialized nations are dotted with abandoned minepits and quarries, with tailings and dredge containment areas. In temperate climates, most eventually fill with water. Post-mining land usage of these ponds and pit lakes has not been planned in the past because of little interest by the private sector. In the past 20 years, as reclamation rules and regulations have become more stringent, mining companies and public agencies have emphasized their potential beneficial uses, which include wildlife habitat, aesthetics, and recreation (Svedarsky & Crawford 1982; Brenner et al. 1987; Acott 1989; Constantino 1989; Amell & Eastwick 1996); drinking water sources (Minnesota Pollution Control Agency [MPCA] 1990); landfill (Weaver 1991); lakeshore residential housing (Maneval 1975; Browning 1987); wetland creation for mitigation (Moshiri 1993); and, more recently, commercial aquaculture (Homziak & Lunz 1983; Lunz & Konikoff 1986; Svatos 1986; North Central Region Aquaculture Center 1996).

Minnesota, the land of more than 10,000 lakes, is also blessed with more than 4000 abandoned quarry pits and over 200 deep, abandoned iron ore pits (Buttleman

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1992). In the past 25 years, the iron ore pits have gradually filled with groundwater, forming lakes on the Cuyuna, Mesabi, and Vermillion Iron Ranges in northeastern Minnesota (Figs. 1 & 2). The lakes are typically small, ranging in size from 2–100 hectares, steep-sided, and relatively deep, with maximum depths from 20 m to over 200 m. A few are used for drinking water by iron range communities (MPCA 1990), and about 20 are managed for recreation (Svatos 1986; Iron Range Resources and Rehabilitation Board 1988; Pierce & Tomcko 1989). Unlike many pit lakes located in basins where either coal or non-ferrous metals were extracted (Miller et al. 1996), the water quality of most of the iron ore pit lakes typically does not exceed federal standards for drinking water or freshwater aquatic organisms (Pierce & Tomcko 1989; Axler et al. 1992*b*, 1996).

In the late 1980s, studies of the feasibility of aquaculture of salmonids in the mine pit lakes produced estimates that $44 million/year in primary and secondary revenues could be generated (Colt et al. 1987, 1989). In 1988 a commercial enterprise (Minnesota Aquafarms, Inc. [MAI]) obtained ownership of five pit lakes and began intensive production of chinook salmon in net-pens in two of the abandoned mine pit lakes (Twin City–South and Sherman) near Chisholm, Minnesota (Figs. 1–3).

Disagreements immediately developed regarding the potential effects of this activity on the current and potential future beneficial uses of these lakes and on the regional aquifer (Axler et al. 1992*a*, 1992*b*, 1995, 1996). The controversy was particularly emotional regarding the potential for long-term water quality degradation in the nearby Fraser pit lake, which was the drinking water source for the city of Chisholm (population ~5000) and which lies immediately to the north of the Sherman Pit and to the northeast of Twin City–South (Figs. 1 & 2).

Although aquaculture has been the fastest growing sector of agriculture in the United States in the past 10–15 years, as the industry has expanded so have concerns about the effects of aquaculture effluent on water quality (Axler et al. 1992*b*, 1996; Folsom & Sanborn 1993; Rubino & Wilson 1993; Batterson & Piedrahita 1996; Costa-Pierce 1996). Net-pen aquaculture is the most economically feasible method of intensively raising fish such as salmonids (Beveridge 1987). Proposed or current net-pen aquaculture sites include fjords and coastal areas in Sweden, Norway, Finland, Chile, Canada, Maine, Alaska, California, and Washington State (Eklund 1984; Cline 1989; Ackefors & Enell 1990; Parametrix 1990; Bergheim et al. 1991); Scottish sea lochs and lakes (Beveridge & Muir 1982; Brown et al. 1987); dugout ponds in South Dakota (Roell et al. 1988); southeastern U.S. reservoirs (Hays 1980); dredge spoil basins (Homziak & Lunz 1983; Lunz & Konikoff 1986); and abandoned quarries and mine pit lakes (Svatos 1986; Axler et al. 1992*b*; North Central Region Aquaculture Center 1996). In intensive systems, organic enrichment comes primarily from uneaten food and fish feces dispersed to the surrounding water. In a commercial operation, as much as 30% of dry feed (including fines) remains uneaten (Rosenthal et al. 1988), and 25–30% of the consumed food is digested as feces (Nature Conservancy Council 1990). Numerous studies (Enell 1982; Leonardsson & Näslund 1983; Brown et al. 1987; Nature Conservancy Council 1990; Axler et al. 1996) have demonstrated hypolimnetic and sediment anoxia due to organic enrichment from intensive aquaculture. These anoxic conditions may also enhance the exchange of nutrients,

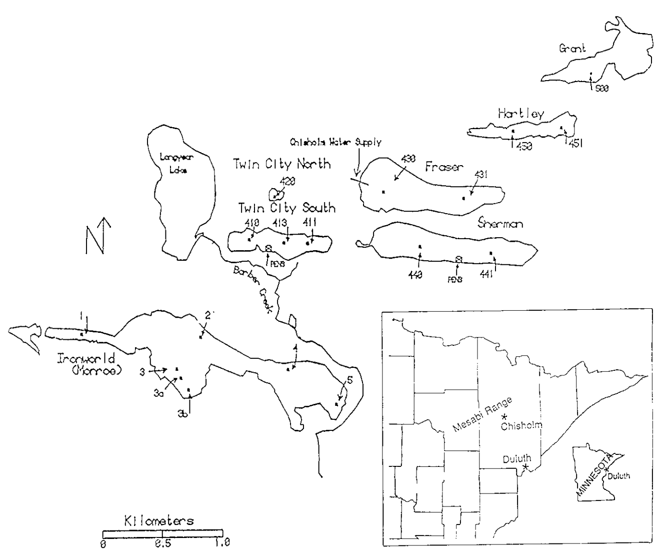

Figure 1. Map of Iron Ranges and sites near Chisholm, Minnesota, showing sampling stations and the geographic relationships between the stations. Grant pit lake is located ~5 km east-northeast of Fraser pit lake.

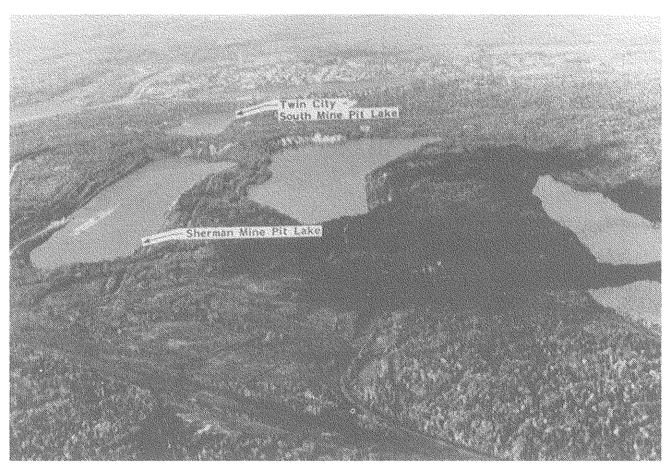

Figure 2. Mine pit lakes near Chisholm, Minnesota, U.S.A. The top of the photo points due west. Fraser pit lake lies immediately to the north (right) of Sherman pit lake.

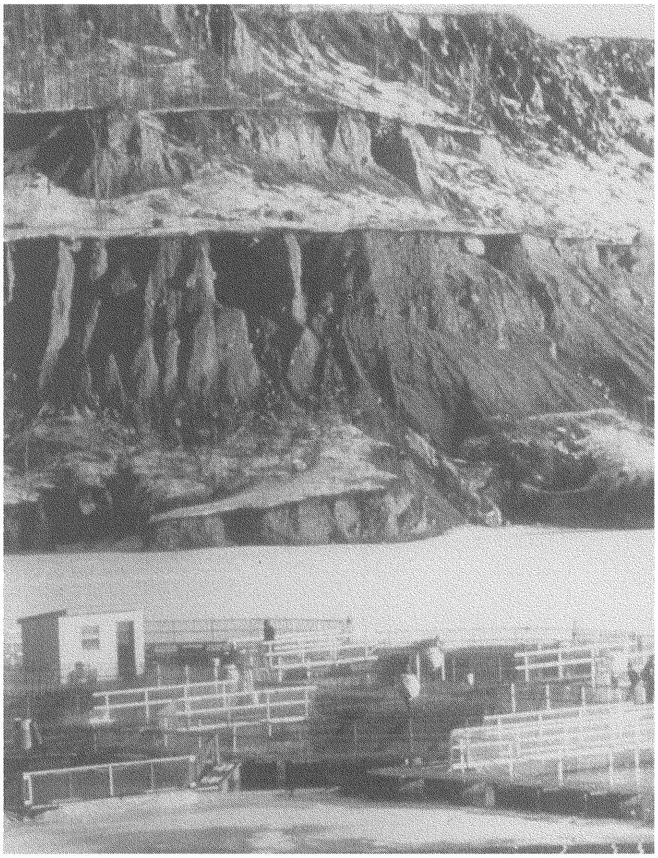

Figure 3. Aquaculture facility at the Twin City–South mine pit lake showing the north wall. (Photos courtesy of Intertribal Business Network.)

particularly phosphorus, between the sediments and the overlying water (Nürnberg 1988).

In September 1992, in response to violations of some provisions of the permit for MAI's National Pollutant Discharge Elimination System/State Disposal System, a stipulation agreement was negotiated between the company and the MPCA (1992). In addition to setting interim in-lake water quality limitations and requiring a waste collection and treatment system in the Sherman pit lake, the company was required to conduct a complete restoration of the Twin City–South pit lake within three years to demonstrate that it was economically feasible and that the nutrient enrichment effects of intensive aquaculture on the pit lake were reversible. The basic restoration plan was to (1) turn off the "load" (i.e. cease aquacultural activity); (2) aerate and vertically circulate the water column to prevent hypolimnetic anoxia and to prevent (potential) surface blooms of algae; (3) seal sediment organic matter (sediment oxygen demand) by accelerating basin wall erosion of inorganic till, (4) precipitate water column phosphorus by adding alum/aluminate, which would also contribute to preventing the release of phosphorus from the sediments.

Fish feeding was to be terminated before 1 May 1993, and fish were to be removed before 1 July 1993. MAI was also required to form an independent technical peer review panel of consultants to assist with the development of the restoration plan. The stipulation agreement mandated a stepwise approach to restoring Twin City–South water quality. It limited the first year's restoration activities to artificial circulation and aeration and prohibited other measures, such as applications of chemical precipitants, until the second year (1994). The effectiveness of the fallowing would be evaluated against the following assumptions about the baseline trophic state of TC-S: total phosphorus (TP) $\leqslant$10 ug/L (growing season mean, 1 May–30 September); chlorophyll *a* $\leqslant$3 ug/L (monthly mean, 1 May–30 September); secchi transparency $\geqslant$3 meters (growing season mean, 1 May–30 September); dissolved oxygen (DO), no presence of biogenic meromixis or anoxia (i.e., DO $<$ 0.1 mg/L) in the lower half of the hypolimnion, excluding the lowest 3 m above the bottom.

The Natural Resources Research Institute has conducted studies since July 1989 on the limnology of the aquaculture pit lakes, including their response to aquacultural nutrient enrichment and the potential effects of this enrichment on nearby surface and groundwater. Water quality information and analyses regarding aquacultural impacts on the Sherman pit lake and on nearby nonaquaculture "reference" pit lakes have been reported by Axler et al. (1992*a*, 1992*b*, 1995, 1996). We report here on the changes in water quality and sediments in the Twin City–South mine pit lake following the termination of all aquaculture activities. We also describe the potential restoration methods mandated by the state and suggest potential improvements in this process. Demonstration of environmentally effective and affordable restoration methods is a prerequisite for continued aquacultural reclamation of these or other mine pit lakes in Minnesota, or of similar lakes elsewhere.

Study Sites

Tables 1 and 2 summarize the morphometry, general chemical characteristics, and aquacultural history of the study lakes. Twin City–South (TC-S) was used for intensive net-pen salmon farming from the summer of 1988 until the spring of 1993. The Sherman pit lake was used for aquaculture from 1989 through 1995, although production was phased out through the 1995 field season. All aquaculture operations in TC-S were terminated, including fish feeding (by 1 May 1993) and penned fish (by 1 July 1993), as mandated by the Minnesota Pollution Control Agency. TC-S was artificially mixed and aerated in the summer, beginning in 1989 and continuing through 1993. Mixing intensity was substantially increased in June 1991 and maintained from summer until

Table 1. General morphometric and chemical characteristics of the Twin City–South mine pit lake.[a]

Morphometry		*Chemistry*	
Area (ha in 1989)	28.3	EC (μS/cm)	450
Volume ($m^3 \times 10^6$)	8.0	pH	7.1–8.4
Maximum depth (m)	64.0	Alkalinity (mg $CaCO_3$/L)	164
Mean depth (m)	28.0	Hardness[b] (mg $CaCO_3$/L)	222
Relative depth (%)	10.0	SO_4 (mg/L)	60
Midsummer mixed layer (m)		Cl (mg/L)	17
• Artificially mixed		Ca (mg/L)	51
1989–1992	15–45	Mg (mg/L)	23
1993	~70	Na (mg/L)	9.5
• Natural stratification		K (mg/L)	3.5
1994–1995	5–7	Fe (mg/L)	<0.1
Annual mixing (fall and spring)	complete	Fe (anoxic hypolimnion)	0.1–1.1

[a]Volumes and depths are for 1988 and have increased approximately 2% each year. Chemical values are averaged over all depths and dated from 1989 to 1993. Differences between depths were minor for major ions.
[b]Calculated as H = 2.497 [Ca] + 4.118 [Mg].

fall in subsequent years. The mixers and aerators were turned off for several weeks in August 1993 to enable estimation of oxygen depletion rates.

Fraser pit lake is the raw drinking water supply for Chisholm, Minnesota (population ~5000) and was not used for intensive aquaculture (Axler et al. 1996). It lies immediately adjacent to the aquaculture pit lakes (Figs. 1 & 2). Grant pit lake, approximately 5 km east of the aquaculture lakes, was sampled as a second reference lake, having no aquaculture from 1989 to 1992. The Ironworld Pit lake, formed in the Monroe pit, lies immediately to the south of the Sherman and TC-S aquaculture pits on the other side of State Highway 169.

General Hydrology and Limnology

The Mesabi Range mine pit lakes that are the focus of this study were formed following the cessation of open-pit iron ore mining in the early 1970s. Underground mining activities, which extracted higher-grade ore, had been carried out earlier within many of these pits. When these were exhausted and taconite open-pit mining became feasible in the 1950s, the mines required continual dewatering on the order of 8000 L/minute which lowered the surrounding water table to as much as 130 m below the premining water table (MAI 1993). The bedrock in the study area is heterogeneous and fractured from several decades of previous mining activities; groundwater movement is influenced by nearby (<10 km) dewatering of active taconite iron-ore pits (Short-Elliot-Hendrickson, Inc. 1990; MAI 1993).

During the current filling phase of these pit lakes, the gradient of water movement is into the pits; in fact, lake levels have risen as much as 1 m annually in recent years. After pit lake levels stabilize in 10–40 years (Short-Elliot-Hendrickson, Inc. 1990; MAI 1993), however, some migration of solutes may occur depending upon the direction of groundwater flow and concentration gradients. Water budget estimates for TC-S indicate that groundwater represents about 23–70% of annual inputs. Assuming that surface discharge will occur after the lake is full, the estimated hydraulic retention time would be on the order of 42–108 years (Yokom 1996; Yokom et at. 1997).

The pit lakes are moderately hardwater, alkaline lakes with lower levels of phosphorus and higher levels of nitrate than those of natural lakes in the region (Axler et al. 1996). The major ion characteristics are very similar to those of the Biwabik Iron Formation Aquifer, which is the source of much of the water seeping into the pits (Anderson 1986). Calcium and magnesium are the predominant cations and bicarbonate and sulfate the major anions. Although chloride levels are low, large differences between adjacent pits provided a useful tracer for showing that potential water movements between pits were minimal (Axler et al. 1992*a*, 1996). High

Table 2. Annual loading of fish food to Twin City–South mine pit lake, food concentrations as total solids, total phosphorus (TP), and total nitrogen (TN) normalized to whole-lake volume.

Year	*Food (metric tons/year)*	*Food Solids (mg/L)*	*TP (mg/L)*	*TN (mg/L)*
1988	66.7	8.34	0.18	0.62
1989	219.4	27.4	0.42	2.02
1990	299.7	37.5	0.41	2.68
1991	361.1	45.1	0.39	3.24
1992	275.2	32.5	0.37	2.34
1993	6.38	0.73	0.01	0.06
1994	0	0	0.00	0.00
1995	0	0	0	0

ratios of nitrogen to phosphorus (either as total or inorganic) and midsummer algal bioassays (Axler & Yokom, unpublished data) indicated that algal productivity during the growing season was likely to be limited by available phosphorus. Occasional reductions in light penetration due to high turbidity from direct surface runoff following spring snowmelt and rainstorms, and the deep vertical mixing caused by the aerators and mixers, have prevented the development of algal blooms typically associated with high rates of nutrient loading (Axler et al. 1996).

Ice typically forms by early December and persists until late April. Thermal stratification is rapid and pronounced, with a thermocline in the range of 5–7 m throughout the summer growing season. Although there are spring and fall mixing periods, some mine pit lakes do not mix to the bottom every year because of their great depth relative to their surface area and because they often lie in sheltered, steep-sided basins. This morphometry results in incomplete reoxygenation of hypolimnetic water and/or incomplete redistribution of solutes (Axler et al. 1996). As a consequence of incomplete mixing, the oxygen demand of inundated terrestrial soil and vegetation, and perhaps occasional historical inputs of domestic wastewater from neighboring towns, some mine pit lakes may "naturally" develop anoxic hypolimnia (Pierce & Tomcko 1989). But most water in the deeper lakes is expected to have relatively high levels of dissolved oxygen (>5 parts per million) and relatively cool temperatures, both of which are prerequisites for salmonid culture and growth.

Methods

Twin City-South mine pit water was sampled from one or two midlake deepwater sites at approximately monthly intervals during the growing season (May-October) from July 1989 through 1994, and twice in 1995. Permit data for the National Pollutant Discharge Elimination System were also available from Minnesota Aquafarms, Inc. (MAI 1994), and were incorporated into our database after extensive QA/QC (Axler et al. 1992*a*). Early- and late-winter samples were also collected from 1989 to 1994.

Surface water composites (0–2 m) were collected with a polyvinyl chloride (PVC) tube 5 cm in diameter and 2 m in length; discrete depth samples from 4–6 depths to 1 meter from the bottom were collected with an opaque PVC Van Dorn bottle. Temperature, dissolved oxygen, pH, and conductivity were measured with a Hydrolab Surveyor II water quality analyzer. Water transparency was estimated with a standard 20-cm secchi disk and light attenuation with a Licor PAR quantum sensor. Water chemistry analyses followed standard methods (American Public Health Association 1989; Owen & Axler 1991; Ameel et al. 1993). Chlorophyll was analyzed both spectrophotometrically and fluorometrically from 90% acetone extracts (Axler & Owen 1994). QA/QC procedures were rigorous and based on a detailed field and laboratory manual that is revised annually and certified by the State of Minnesota for federal Safe Drinking Water and Clean Water Act procedures. Volumetric hypolimnetic rates of oxygen depletion (VHODs) were calculated from changes in volume-weighted hypolimnetic dissolved oxygen concentrations during summer stratification.

Surficial bottom sediment was collected during 1989–1993 with a stainless steel 15 × 15 cm Ekman dredge. In most cases this provided a reasonably undisturbed sample (evidence being intact chironomid-oligochaete burrows) that was cored in the field with cut-off 60-ml polypropylene syringes (2.7 cm diameter). A limited number of samples was collected using a KB drop corer and two custom-made piston corers. An independent coring effort was contracted by MAI in February 1994, which used a CO_2/alcohol freeze corer (Wright 1980) to provide estimates of sediment accumulation and recovery in the immediate vicinity of the net-pens. Sixteen cores were collected to provide synoptic coverage of the lake bottom, of which eight were sectioned for chemical analyses as per Engstrom and Wright (1984).

Duplicate or triplicate sets of sediment traps made of 3.8 × 30–cm PVC pipe (aspect ratio 8:1) were suspended about 2 m off the bottom at mid-lake sites along the main east-west channel of the lake. No preservative was used. Organic matter was estimated as ash-free dry weight, organic carbon by CHN analysis after fuming with HCL, percent phosphorus (P) by extraction of ash residue in 6N HCl, and bioavailable phosphorus by extraction in 0.1N NaOH (details in Owen & Axler 1991). The extraction efficiency was checked against a certified U.S. Environmental Protection Agency reference sediment sample with a P content of 0.1%, and our mean recovery was 112 ± 6% (mean ± SD, $n = 6$). The CV for material from pairs of sediment traps was typically less than 10% for all parameters.

Sediment oxygen demand and sediment P release for experiments 1 and 2 were estimated by means of microcosms containing sediments collected from areas not directly influenced by particulate wastes from the net pens. Sediments were collected with an Ekman dredge, homogenized, and analyzed for total phosphorus and organic matter. Each microcosm consisted of a 1-L mason jar filled with GF/C filtered lake water and a petri dish 6 cm wide and 1.5 cm deep used to hold sediment. The resulting sediment surface area was 0.0022 m^2, and the sediment ratio of surface area to volume was 0.029/m. Microcosms were sealed without a headspace and kept in a dark incubator at 10°C. Sampling was minimized to limit microcosm disturbances. Dissolved oxygen was measured with an air-calibrated YSI BOD probe, and ortho-P was measured as described above. Following each sampling, microcosms were resealed without a headspace with GF/C filtered lake water and returned to the incubator. Sediment oxygen demand and P release rates were estimated by linear regressions of temporal concentration changes.

Sealing and capping experiments with concentrated waste sediment collected from the immediate vicinity of the pens (experiment 3) were also performed. Approximately 10 cm of sediment were overlaid by 3 L of lake water in duplicate PVC microcosms 1–1.25 m tall and 10 cm in diameter. A 50-g/m^2 dose of alum was later added, along with sufficient NaOH to achieve a final pH of 6.0. Treatments involved adding varying amounts of sand to cap the organic sediments. The surface water was capped with 3 cm of mineral oil to impede the diffusion of atmospheric oxygen. Linear regression analyses of temporal changes in phosphorus mass data were conducted for each sediment-water microcosm to estimate anoxic sediment phosphorus release rates. Similar statistical analyses of dissolved oxygen mass data were performed to estimate rates of sediment oxygen depletion prior to the establishment of anoxia.

Fish food loading to the net-pens was continuously recorded by MAI. We estimated total P loading using the food phosphorus content reported by the feed company and certified by MAI (1994). On several occasions we corroborated these percentages by determining P content by the method for sediment described above. Nitrogen content was estimated as 7.48% of dry weight and periodically determined directly with a Leco CHN analyzer calibrated against acetanilide standards.

Results and Discussion

Nutrients

Water quality rapidly returned to a condition approximating baseline (the pre-aquaculture data set is limited) following the cessation of aquaculture activities. By the end of the 1993 growing season, total water column phosphorus declined to below Summer 1989 levels, (Fig. 3). Surface values, upon which the regulatory standard is based, were less than 10 ug P/L by the end of the summer of 1993, and by early 1994 the entire water column had levels below 10 ug P/L (Fig. 4). The rate of whole-lake P loss from the water was 44 ug P/L/year, calculated from April 1993 through November 1994 P budgets. This value is similar to the mean rates of decrease of surface water total P, 43 ug P/L/year (1992 to 1993) and 37 ug P/L/year (1993 to 1994), calculated from Figure 4. The rapid recovery of the lake's P budget, despite no losses from outflow, was the result of several factors: (1) less than 50% of the water column phosphorus was particulate. Aquacultural P loads are comprised primarily of uneaten food pellets and feces, both of which settle relatively rapidly (Rosenthal et al. 1988). (2) The lake has a high relative depth, which minimizes transport from bottom water and surficial sediments, even when artificially mixed to great depth. (3) unlike most agricultural nutrient runoff scenarios, the source was entirely within the water body. Therefore, when feeding was stopped in May 1993, the external nutrient loads were effectively reduced to zero almost instantaneously (the fish in the pens were required to be totally removed by July 1993), (4) The "natural watershed" nutrient load was extremely low because the basin was small and the substrate was mostly glacial sediment and mine tailings.

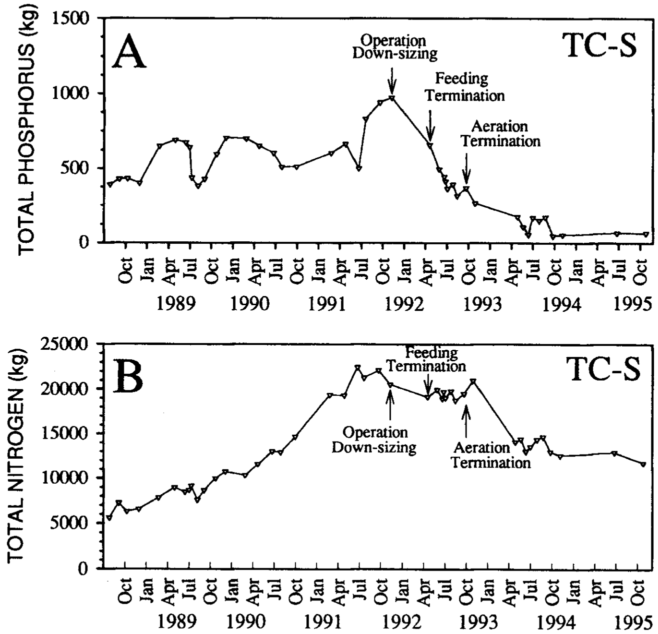

Figure 4. Whole-lake (volumetrically integrated) total phosphorus (TP) and dissolved inorganic nitrogen (DIN = ammonium + nitrate + nitrite, as nitrogen) budgets for Twin–City South mine pit lake. DIN comprised over 95% of the total nitrogen in the water column at all depths.

Nitrogen levels also declined, although more slowly than those for P (Figs. 4 & 5). Virtually all of the nitrogen present in the water column in 1993 and 1994 was in dissolved inorganic form (nitrate + nitrite + ammonium), as for previous "aquaculture" years (Axler et al. 1992*a*, 1995, 1996; MAI 1994), but more than 95% of the dissolved inorganic nitrogen (DIN) was in the form of nitrate after April 1993. This condition was predicted previously (Axler et al. 1992*a*, 1992*b*) because the major source of ammonium was fish excretion and mineralization of organic wastes, both of which were eliminated in the spring of 1993. Residual water column ammonium was nitrified relatively rapidly to nitrate by

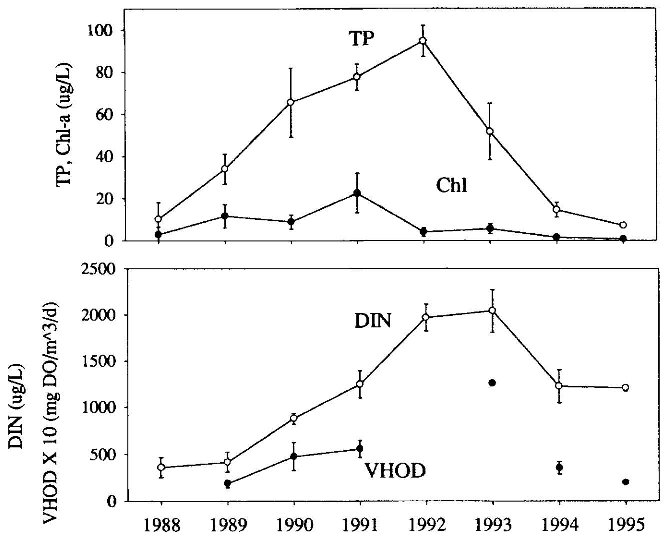

Figure 5. Twin City–South summer (May-September) surface water values for total phosphorus (TP), dissolved inorganic nitrogen (DIN = ammonium + nitrate + nitrite, as nitrogen), chlorophyll *a*, and the volumetric hypolimnetic oxygen depletion rate (VHOD). 1988–1995. VHOD for 1992 could not be estimated because of intensive aeration; VHOD for 1993 was based upon oxygen changes during a 3-week period in August when the aeration-circulatian system was temporarily turned off; VHOD for 1995 (no aeration and circulation) was based on dissolved oxygen profiles measured on only two dates (June and October).

aerobic, nitrifying bacteria. Some must have been assimilated by algae, but in 1993 algal production was low due to light limitation from the deep mixing by MAI's aeration system throughout most of the growing season (Axler et al. 1996). On a whole-lake basis, there was little loss of DIN during the first year of fallow, and then an average loss rate of 0.43 mg DIN/L/year from July 1993 to November 1994 (Fig. 4). The mean loss rate from surface water from 1993 to 1994 was 0.81 mg DIN/L/year (using Fig. 5). The loss rate for surface water DIN during recovery has thus far been similar to its rate of increase during periods of peak aquacultural production. The range of estimated DIN decrease is consistent with estimated algal uptake in a moderately productive lake such as TC-S in 1993 and 1994. For a mesotrophic rate of phytoplankton primary productivity of 250–1000 mg carbon/m^2/day (Wetzel 1983, Table 15-9), a carbon-to-nitrogen (C:N) uptake ratio of 7 (approximate Redfield ratio, in g/g) and a 150-day growing season, this would imply an inorganic nitrogen assimilation of 0.19–0.77 mg nitrogen/m^3/growing season, which encompasses the observed loss rates. This suggests that a major fraction of the assimilated N from the 1993 growing season was sedimented prior to that season.

In the context of public health, although the current nitrate concentration at TC-S ($\sim$1.3 mg N/L in late October 1995) remains approximately 0.9 mg N/L above the pre-aquaculture baseline, it is almost 9 mg N/L below the state and federal drinking water standard of 10 mg [NO_3-N]/L (U.S. Environmental Protection Agency 1995) and should continue to decline.

Algae

Algal blooms have always been infrequent in the aquaculture pit lakes because of light limitation from vertical mixing (Axler et al. 1996). The seasonal averages for 1993 and 1984 remained 5 ug/L or less as expected (Fig. 3). Enhanced phytoplankton growth was previously predicted during the first few years of fallowing after artificial mixing was discontinued because of initially high nutrient levels (Axler et al. 1992*a*). But the continuation of artificial vertical mixing in 1993 prevented this from occurring, and by 1994 the algae were presumably limited by available P, as in the baseline condition when P was low and N relatively high (Figs. 4 & 5).

Dissolved Oxygen

Fallowing, along with one summer of aeration and mixing, caused a rapid return to pre-aquaculture hypolimnetic oxygen conditions (Fig. 6). Midsummer dissolved oxygen (DO) levels of $\geqslant$5 mg O_2/L persisted near the bottom (62–65 m) throughout late summer in 1993. Minimum fall DO values remained relatively high in near-bottom water in subsequent years, >3.9 mg O_2/L in November 1994 and > 3.4 mg O_2/L in October 1995, despite no artificial mixing. During intensive aquaculture (1990–1992), hypolimnetic DO was chronically <3 mg O_2/L, and often <1 mg O_2/L. During much of the summer the DO concentration in the upper mixed layer was also severely undersaturated, with values commonly below 5 mg O_2/L at the surface (Fig. 6). Most of the oxygen demand in the mixed layer during aquaculture (often more than 45 meters deep from artificial mixing) was previously shown to be due to fish respiration (Axler et al. 1993; McDonald et al. 1996). Hypolimnetic oxygen depletion was caused by settling organic wastes in the water column and by sediment oxygen demand. The nitrification of ammonium presumably also contributed to the DO deficit in the water column, although to a lesser extent.

Water column oxygen sinks were almost entirely eliminated soon after the fish were removed. Because algal growth was typically low, the contribution of this source of organic matter to hypolimnetic DO depletion was also low. As described below, the sedimented aquaculture wastes were rapidly buried under a blanket of inorganic material that "naturally" erodes from the basin walls during spring runoff and rainstorms, and from shoreline erosion. As a consequence, the rate

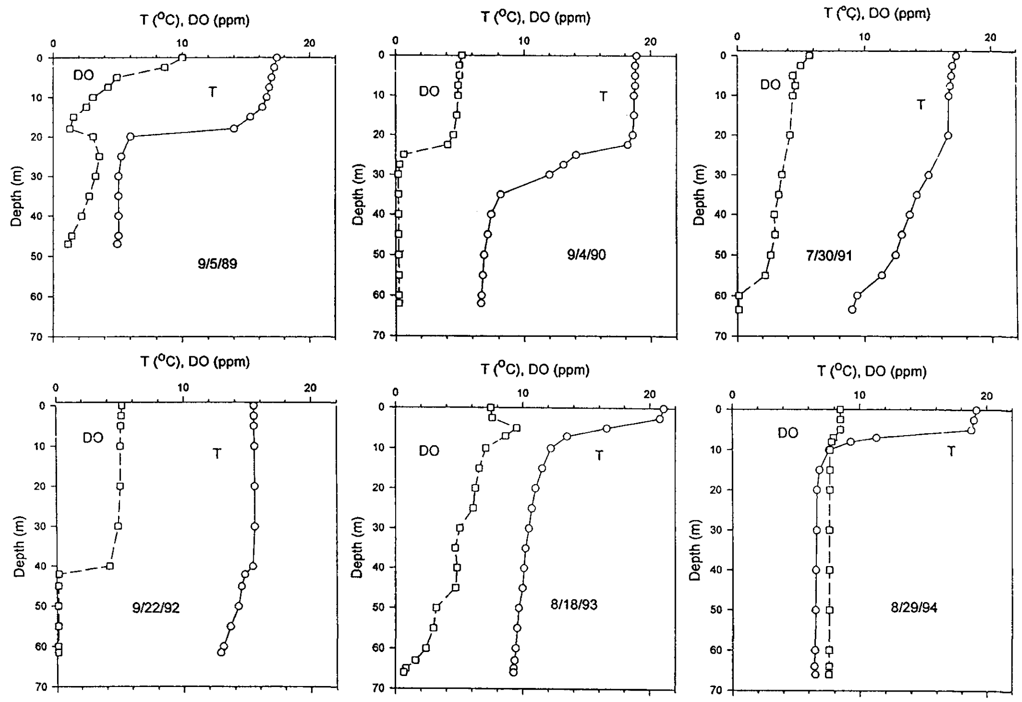

Figure 6. Late-summer vertical profiles of temperature (°C) and dissolved oxygen (ppm) for Twin City–South mine pit lake, 1989–1994.

of volumetric hypolimnetic oxygen depletion (VHOD) decreased significantly in 1994 relative to previous years (Fig. 5). VHODs could not be calculated for 1992 and 1993 because of MAI's vertical mixing. Also, there were insufficient early season data in 1988 to provide an accurate estimation of baseline conditions for TC-S.

Because of the great relative depths of these mine pit lakes, it cannot be assumed that the TC-S hypolimnion was reaerated naturally to 100% saturation each spring overturn prior to aquaculture. The lakes may thermally stratify soon after ice-out, leaving much of the water column initially undersaturated (Axler et al. 1996). This effect is more obvious in the deeper ($Z_{max} > 100$ meters) Fraser pit lake, which was not aquaculturally loaded, where the hypolimnion below 75 m has not had DO levels greater than about 70% saturation since the inception of our studies in 1989; some years it has been below 40% (MAI 1993; Axler at al. 1996). It is also possible that occasional inputs of untreated wastewater from the Chisholm Wastewater Treatment Plant to TC-S from chronic infiltration and inflow problems during high-runoff events has contributed in the past to somewhat higher VHODs in TC-S. Although such events have been a regular occurrence during major summer storms during our studies, the settleable organic matter is probably buried in the same manner as aquaculture wastes (see below).

The Scientific Peer Review Panel, which approved MAI's restoration plan, noted that the majority of the deeper mine pit lakes predisposed them to meromixis (as routinely observed for Fraser pit lake, which is more than 100 m deep). They concluded that meromixis was a natural process unlikely to be affected significantly by aquaculture and not a significant consideration requiring action in the restoration plan.

Trophic status

The Carlson (1977) Trophic State Index (TSI) was calculated because the MPCA uses it to classify Minnesota lakes according to regional patterns and to provide a framework for evaluating the water quality characteristics of specific lakes (MPCA 1990). The index is the mean of three components based on total phosphorus and chlorophyll concentrations and secchi depth: it was calculated for TC-S surface water during the period May through September (Fig. 7; n $\geq$ 22/year, 1988–1994). According to this index, TC-S recovered from a condition during the peak aquaculture years (1989–1992) characterized as "supportive of most beneficial uses, but threatened" to a state "fully supportive of all swimmable and aesthetic uses" (Carlson 1977; MPCA 1990). The mean value for 1994 was 37 ± 2.5 (mean ± 95% C.I., which is well below the ecoregion average of 47 and would generally be characterized as oligo-mesotrophic. Values for nearby, nonaquaculture pit lakes in the region were 29 ± 2 for Fraser pit lake from 1988 to 1994, 36 ± 16 for Grant pit lake from 1989 to

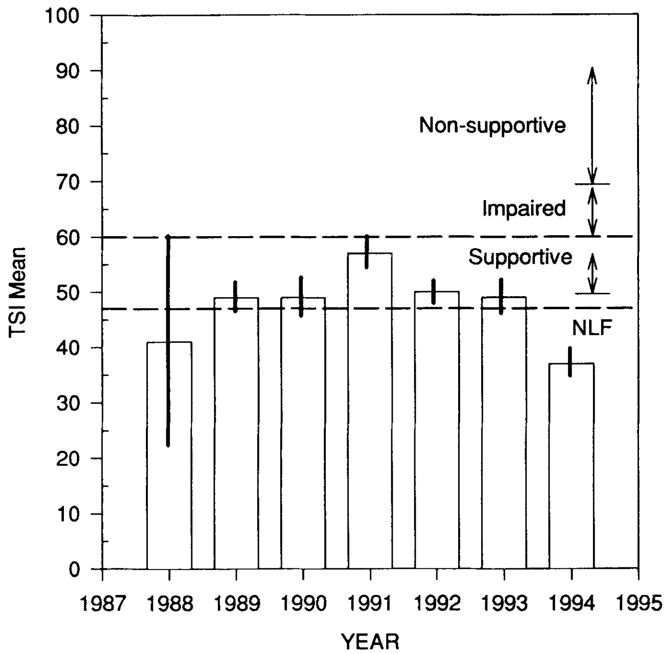

Figure 7. Summary of Carlson TSIs (mean [TSIP/TSIC/TSIS] for May- September) for Twin City–South mine pit lake. Values represent means of pooled data from monitoring programs of the Natural Resources Research Institute and Minnesota Aquafarms, Inc. (for details see Axler et al. 1992*a*). Error bars denoted a 95% confidence interval for each year. NLF identifies the mean value of 47 for intensively monitored lakes in the Northern Lakes and Forests ecoregion (Minnesota Pollution Control Agency [MPCA] 1990). MPCA ranges for beneficial uses: <50, fully supportive of all swimmable and aesthetic uses; 51–59, supportive but threatened; 60–65, partially supportive but impaired; >65, nonsupportive of swimmable and aesthetic uses (MPCA 1990).

1992, and 36 ± 2 for Sherman pit lake during the pre-aquaculture period 1988–1990 (Axler et al. 1998).

Transparency

Previous analyses showed that water clarity changes in TC-S were usually due to silt from shoreline erosion which settled relatively rapidly (days) after winds and runoff subsided (Axler et al. 1996). But occasional increases in algal growth and perhaps particulate wastes from aquaculture were shown to be associated with short-term decreases in secchi depth during previous years. The growing season means for 1993 and 1994 were the highest measured for TC-S since 1988 (Fig. 8). Secchi values in the summer of 1993 occasionally exceeded 10 m, presumably due to low algal productivity. The mean value for the 1993 and 1994 growing seasons was 4.2 m, which is quite transparent compared to most Minnesota natural lakes and in the upper 75th percentile of the state's inventory of lakes (MPCA 1993).

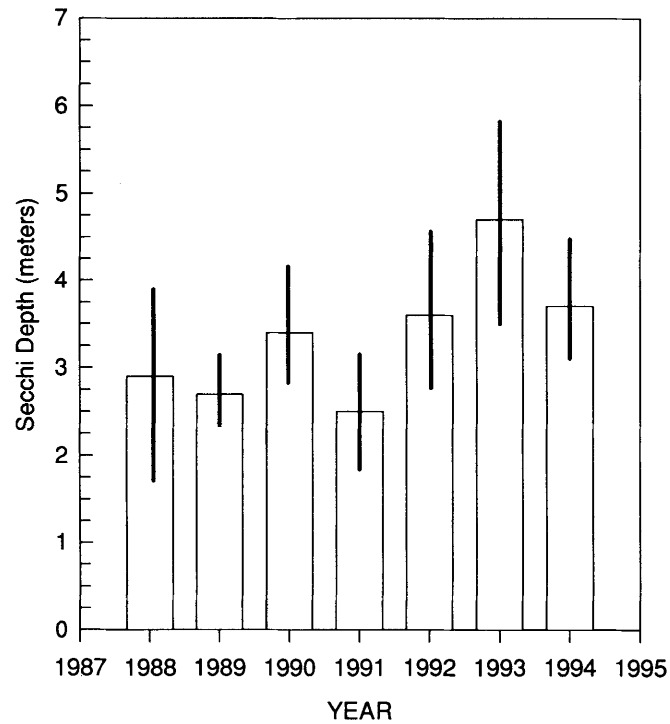

Figure 8. Summer (May-September) secchi depth values (mean ± 95% confidence interval), 1989–1994.

Total Organic Carbon

Although there was no specific target for total organic carbon (TOC) concentration set for TC-S in the stipulation agreement, a value of 5.0 mg/L was set as an interim in-lake limitation for the Sherman pit lake during aquaculture, and it was specified in the state's Aquacultural Development Act (1992) as a standard for water bodies receiving aquacultural waste. The value of 5.0 mg/L was apparently chosen based upon a study of midwestern reservoirs which correlated TOC with total trihalomethane (THM) precursor compounds (i.e., total trihalomethane formation potential; Pope et al. 1988). The concern arises from the potential for trihalomethane formation during the chlorination of drinking water for disinfection, because THMs are suspected carcinogens and federal and state water quality standards exist for *finished* community drinking water supplies. TC-S is not used for drinking water, and so the appropriateness of this standard is questionable. This point is moot because surface water TOC levels during intensive aquaculture (1990–1992) averaged only 3.2 ± 0.2 mg/L (95% c.i.), compared to 2.8 ± 0.2 during recovery (1993–1994) and 3.2 ± 0.7 in 1989. TOC and THM values for Chisholm drinking water, which is drawn from the nearby Fraser pit lake (Figs. 1 & 2) also remained low and essentially unchanged throughout the study period (Axler et al. 1996; Minnesota Department of

Health municipal drinking water supplies monitoring reports for Chisholm, Minnesota).

Higher Trophic Levels

Other than the water quality aspects of phytoplankton growth, we did not investigate changes in the biological communities in TC-S. Zooplankton communities were characterized for the upper and lower water column in 1989 and 1990 and were reported by Axler et al. (1992*a*). Benthic invertebrate communities were depauperate in the main basin, presumably due to sediments of low organic content, high rates of burial, and lack of colonization by deepwater fauna. Post-aquaculture biological surveys were not conducted, but it is likely that the system now has higher biological diversity than before aquaculture, although this may not continue without nutrient subsidies. Introduced minnows—salmonids remaining from aquacultural "ranching" and net-pen escapees—will likely have a strong and highly variable influence on the pelagic and benthic invertebrate communities as ecosystem productivity continues to decline due to nutrient limitation and sediment burial. The productivity and survival of some of these species will probably be limited by the lack of a significant littoral zone. Modern reclamation plans, which must be developed during the pre-mining permitting process, in many cases now require backfilling along some of the mine pit slopes to create a larger, "future" littoral zone after the pit fills with water (Marshall 1983; Hustrulid & Kuchta 1995). But the "natural" biological communities of these highly artificial and hydologically isolated pit lakes were never identified as an issue of concern in the past. Further, from a recreational fishery perspective, inorganic and organic nutrient enrichment from aquaculture would be beneficial. In Alberta, Canada, a recent reclamation plan for a mine pit lake proposed the construction of a wetland at the mouth of the major inflow tributary for the purpose of increasing nutrient inputs to the lake to enhance its fishery potential (Amell & Eastwick 1996).

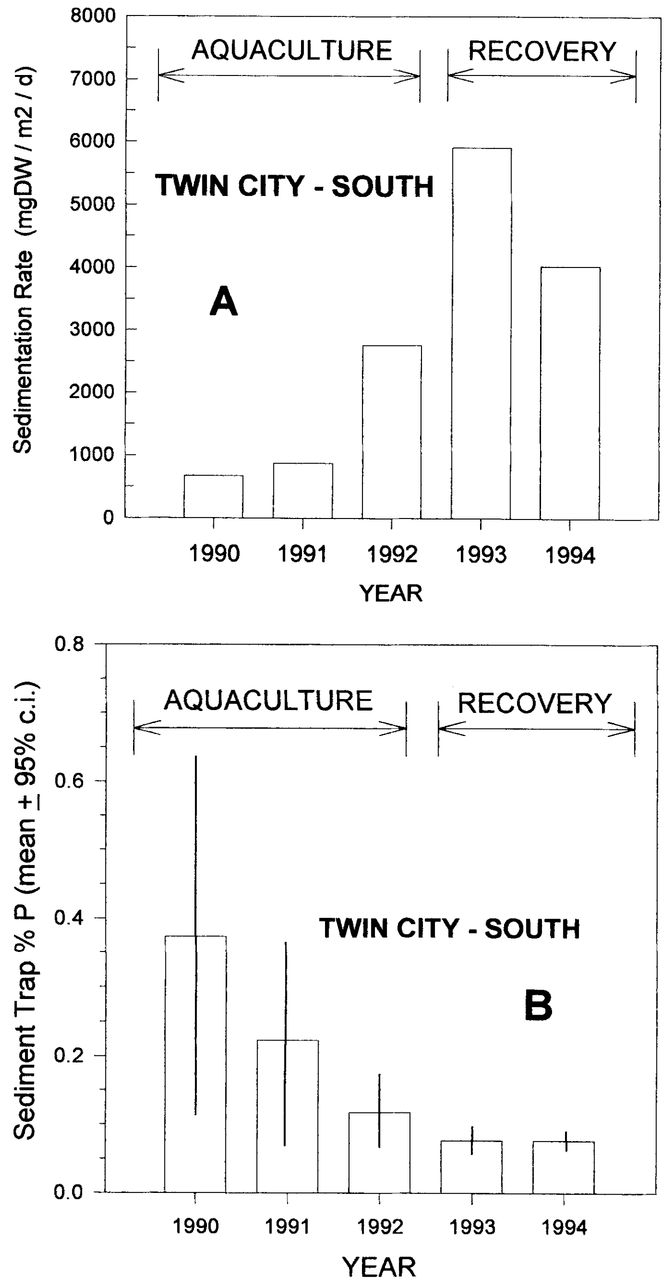

Figure 9. Median annual rate of total sedimentation for Twin City–Sourth mine pit lake, determined with sediment traps (a); mean (±95% confidence interval) concentration of phosphorus in material collected in sediment traps (b).

Rates of Sedimentation

Median annual rates of total sedimentation and the mean annual phosphorus content of the settling seston measured over time by midlake sediment traps are presented in Figure 9. It is apparent that the rate of sediment deposition into the traps was influenced more by meteorological events than by aquaculture or lake management, because the greatest rates were measured in midsummer of 1993 and 1994 after the fish were removed and feeding was terminated. The spring and summer of 1993 were among the wettest on record in the Midwest, and severe rainstorms in July 1993 and July 1994 caused extensive flooding in Chisholm. It is likely that these events accounted for most of the midsummer sedimentation. The phosphorus content of the sedimenting material decreased markedly from 1990 to 1991 and then more gradually through 1994. The dramatic decrease was probably due to improved feeding practices, including screening out fish food fines and

Table 3. Summary of sediment chemical characteristics before, during, and after aquaculture at Twin City–South mine pit lake.*

Sediment/parameter	*Ekman subcores (1989–1993; this study)*		*Freeze-cores (Feb 1994)*	
Reference sediments	Fraser/Grant/Early Sherman (1989–1990)		Deep TC-S (pre-aquaculture layer D)	
%TP	0.048 ± 0.016	(n = 13)	0.047 ± 0.027	(n = 8)
%BAP	12.9 ± 3.6	(n = 11)	~1	(n = 1)
%OM	3.4 ± 1.2	(n = 14)	—	
%OC	0.48 ± 0.40	(n = 14)	0.18 ± 0.07	(n = 8)
Aquaculture sediments	TC-S 1989–1990 (pre-intensive mixing/circulation)		TC-S (subsurface layers *B* and *C*)	
%TP	0.199 ± 0.099	(n = 5)	0.316 ± 0.378	(n = 16 strata)
%BAP	67 ± 38	(n = 5)	82	(n = 1)
%OM	14 ± 10	(n = 6)	—	
%OC	8.0 ± 8.3	(n = 6)	2.4 ± 2.4	(n = 16 strata)
Post-aquaculture surficial	TC-S 1993		TC-S	
%TP	0.034 ± 0.018	(n = 4)	0.114 ± 0.027	(n = 8)
	—		(0.03 ± 0.021)	(n = 3)
%BAP	—		49 ± 6	(n = 2)
%OM	3.8 ± 1.2	(n = 4)	(3.7 ± 0.1)	(n = 3)
%OC	—		0.91 ± 0.28	(n = 8)

*TP, total phosphorus; BAP, biologically available phosphorus; OM, organic matter; OC, organic carbon. The primary source of data for the freeze coring was from a detailed analysis of 16 cores collected in February 1994. Each was sectioned into four layers that were visually differentiated, the uppermost being labeled *A*. Additional surficial layer data from a separate study conducted by MAI in December 1993 with a different freeze corer are reported parenthetically for the post-aquaculture period.

using a low-P diet. The termination of aquaculture in the spring of 1993 resulted in a further, steady decline in the P content of settled material, to a value of about 0.06% (Fig. 9). This is similar to the presumed background of 0.05% found in the non-aquaculture pit lakes (Table 3).

Sedimentation rates were spatially variable, with the highest presumed to occur under the net-pens (corroborated by the freeze coring study in February 1994 [MAI, unpublished report]). Our sedimentation rates represent mid-lake averages but will underestimate the true, basin-wide average proportional to the amount of aquaculture solid waste that was not mixed or resuspended into the main body of the lake. In fact, coring studies also showed a high variability across the bottom of the lake associated with the thickness of individual varves (MAI 1994; Axler et al. 1995; MAI, unpublished data) The resuspension of bottom sediments associated with intermittent changes in the artificial mixing regime may also have contributed to the observed temporal variability.

Sediment Accumulation

Sediment surveys were conducted from 1989 to 1991 in nonaquaculture pit lakes in the immediate vicinity (Fraser, Grant, and Ironworld; see Fig. 1), at the Sherman pit lake prior to intensive aquaculture (1989–1990), and at TC-S after the first year of aquaculture, and then one to three times per year until the end of 1993 after aquaculture had been terminated. Aquacultural loading increased the organic matter content of TC-S surficial sediments and their P concentration (in the vicinity of the net-pens) in the period 1989–1991 (Table 3, Fig. 10). Localized pockets of sedimented aquaculture waste muck in the immediate vicinity of the pens were periodically characterized by noxious-smelling hydrogen sulfide and strong fish odors. Rates of aquaculturally affected sediment accumulation during the initial phase of aquaculture (Summer 1988–October 1990) ranged from 10 to 29 cm/year, as estimated from measurements of the thickness of different-colored strata within subcores taken from the dredge samples. This was generally similar to the rate of approximately 6 cm/year of waste accumulation found for the Sherman pit lake between 1990 and 1991 (Axler et al. 1995) when intensive aquaculture was just beginning (Table 2). The clear pattern of varving in these subcores and occasional undisturbed chironomid and tubificid burrows were taken to indicate that the dredge was not disrupting the integrity of most of the sediment collected, except perhaps for the upper 0.5 cm (Axler et al. 1992*a*). The deposition of "natural basin soils" was estimated to be about 10 cm/year.

Despite aquacultural loading, the maximum concentrations of sediment organic matter and P content remained low compared to those of most natural lakes. The mean P content in the surficial sediment during the

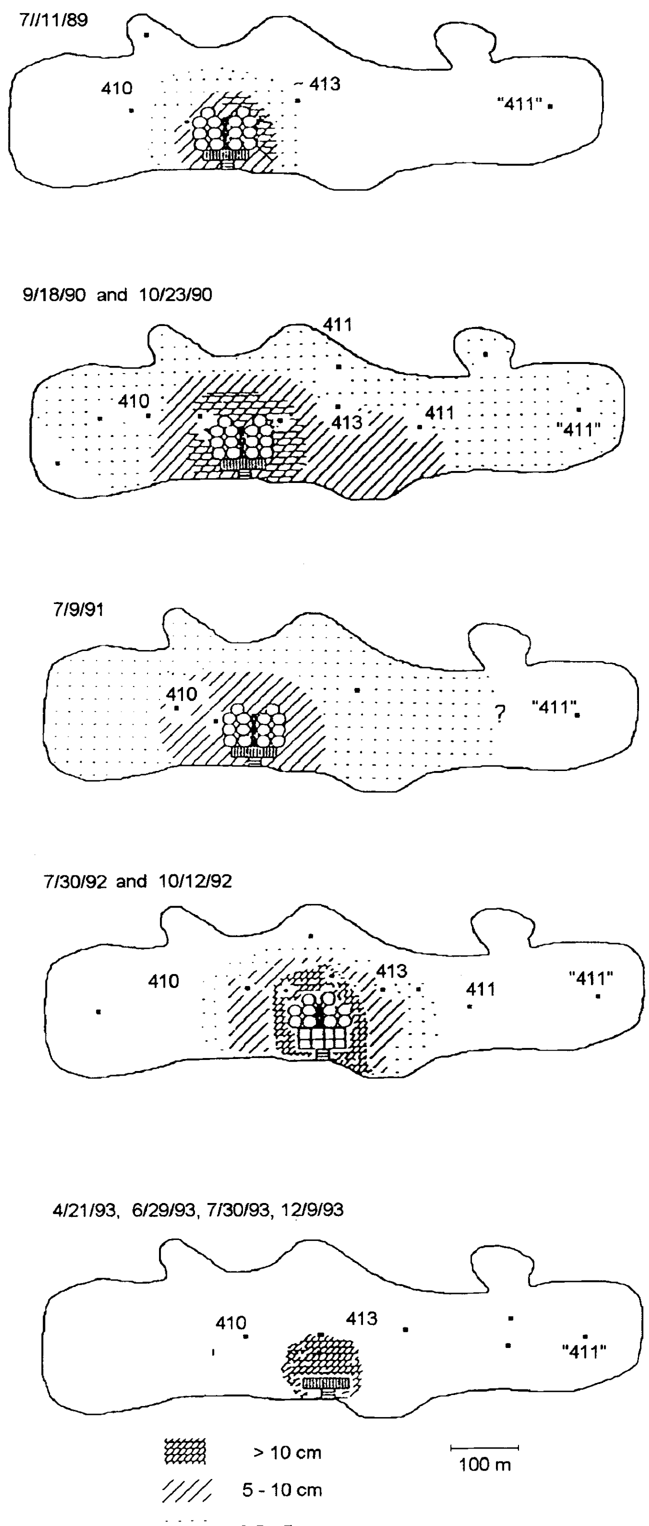

Figure 10. The amount and extent of black, aquaculturally impacted sediment accumulation in Twin City–South pit lake, 1989–1993.

period of most intensive aquaculture was about 0.2%, which is approximately seven times lower than the average P content of fish food (Table 3; MAI 1994). This suggested significant dilution of aquaculture wastes by inorganic, low-P till being deposited from the highly erodible basin walls during runoff events and from erosion by wave action. Layers of oxidized, low-P, and low-organic-matter sediment, which were visually obvious by their orange-red, gray, or tan color, overlaid black layers of aquaculture waste (details in Axler et al. 1995). This pattern also suggested that "natural" burial of these wastes by erosion of the basin walls might provide a mechanism for eliminating the oxygen demand and potential phosphorus release of aquaculturally affected sediments.

Although aquacultural inputs to TC-S were similar from 1990 to 1992, as estimated from total feed load rates (Table 2), the pattern of "muck" accumulation changed after aeration and circulation intensity increased in late 1990. Black material with elevated organic content persisted near net-pens but was sparser elsewhere in the basin. Also, there was no H_2S odor after 1990, even in sediments collected near the pens. Presumably, MAI's increased aeration and circulation at TC-S in late 1990 and 1991 were responsible for reducing the intensity of hypolimnetic anoxia and eliminating noxious odors from surficial sediments, even near the pens, by midsummer of 1991. The dark black color, presumed to be sapropel from reduced iron, was also greatly reduced by the summer of 1991 (Fig. 10). As for TC-S, intensive aeration and circulation at the Sherman facility also prevented noxious odors except directly beneath the pens.

Samples from both ends of the lake and from the central, deep site 413 showed evidence of aquacultural sediment being buried by "natural" inputs of erodible, elastic material from the basin walls. In particular, an orange, red, gray, or tan stratum of 4–12 cm of relatively inorganic mud overlay black material at mid-lake sites 413 and 411 in 1992 (Fig. 10; Axler et al. 1995). The 1993 surveys showed further "improvement," which is presumed to represent the absence of aquacultural loading in concert with continued basin erosion. The upper Midwest also had record or near-record precipitation during the spring and summer of 1993. Aquaculturally affected sediments were buried under at least 4–5 cm of inorganic orange mud at the three main sites. These results were corroborated by the 1994 freeze-core study described above (Table 3; MAI, unpublished report). The analysis showed that aquaculture sediments were covered by erosional deposition throughout the basin and estimated a post-aquaculture sedimentation rate of 3.7 cm/year. The organic matter, carbon, and phosphorus content estimates for baseline and aquaculturally deposited sediments were also in close agreement with our data (Table 3). This suggested that burial by sediment with low levels of P and organic matter produced by basin erosion would probably be effective in mitigating sediment oxygen consumption and phosphorus release after the termination of aquaculture; this formed the basis for Restoration Option II-G (Appendix 1).

Nutrient Inactivation using Precipitants

The lake restoration plan developed by MAI proposed to add chemical precipitants or coagulants to accelerate the removal of P from the lake following the cessation of feeding and removal of fish. Aluminum salts, ferric chloride, and calcium salts have all been used with some success in lake restoration projects either to precipitate dissolved and particulate phosphorus, algae, and silt from lake water columns or to reduce the rate of phosphate release from sediments to overlying water (North American Lake Management Society 1990; Cooke et al. 1993). Calcium additions would have been extremely expensive because of the large volume of the lake, in addition to requiring a prohibitively high pH (~9) to be effective. Ferric chloride was not allowed by the state, despite the fact that TCS was created from an iron-ore excavation. It also presents handling problems because of its acidity (MAI 1993), and the efficacy of ferric hydroxides in "sealing" sediment P might have been limited by the development of anoxia and reducing conditions in the water overlying surficial sediments (Cooke et al. 1993).

A mixture of alum (liquid aluminum sulfate) plus sodium aluminate was ultimately recommended to provide a pH-buffered treatment that would effectively scavenge phosphorus from the water column and prevent release of P from the sediments, independent of hypolimnetic anoxia (MAI 1993). Microcosm experiments demonstrated the potential for an alum treatment to dramatically reduce the rate of P release from anoxic TC-S bottom sediments (Table 4). Also, a previous surface application of alum in July 1990 showed some success in reducing water column P concentrations. The addition was rather light compared to recommendations in the literature in terms of both volumetric and areal application rates (Table 5) because of concern for the inventory of sensitive salmonids that resided in net-pens in the upper 15 m of the lake. But total P was reduced by ~44 μgP/L over the whole lake over a period of approximately 2 months—despite continued rates of fish food loading. Dissolved aluminum levels returned to baseline within nine days of treatment, and there were no significant changes in pH or alkalinity even though the acidic alum was not buffered (Axler et al. 1992*a*).

Table 4. Microcosm experiments simulating the effects of alum and silt or sand on the rates of phosphorus release and oxygen uptake by Twin City–South aquaculturally impacted bottom sediments.*

Treatment	*Temperature (°C)*	*P release from anoxic sediment*		*Oxygen demand of oxic sediment (prior to anoxia)*	
		mg/m²/day	*reduction*	*mg O_2/m²/day*	*reduction*
Experiment 1 (OM = 9%):					
control	10	17	0%	236	—
+alum = 25 g/m²	10	−0.95	>99%	—	—
+1 cm silt/clay cap	10	0.57	97%	—	—
+1cm Bentonite	10	7.4	56%	—	—
Experiment 2 (OM = 6%):					
control	10	—	—	379	—
+1 cm silt/clay	10	—	—	244	36%
+5 cm silt/clay	10	—	—	<67	>82
Experiment 3: Lake Sediment (OM = 55%):					
Alum (50g Al/m²) + Sand (variable thickness)					
No sand, no alum	22	14.4	0%	N/A	N/A
No sand	22	<0.76	>95%	821	0%
0.6 cm sand	22	<0.76	>95%	619	24%
1.3 cm sand	22	<0.76	>95%	372	55%
2.5 cm sand	22	<0.76	>95%	120	85%
5.0 cm sand	22	<0.76	>95%	176	79%
10.2 cm sand	22	<0.76	>95%	161	80%
20.3 cm sand	22	<0.76	>95%	205	75%
5.0 cm sand (no sediment)	22	<0.76	>95%	47	N/A
10.2 cm sand (no sediment)	22	<0.76	>95%	90	N/A
Lake sediment only	4	1.28	0%	1.0	0%
Sand only (20.3 cm)	4	0.46	64%	0.74	26%

*OM, % organic matter; limit of detection denoted by <.

Table 5. Comparison of the magnitude of the 2 July 1990 Twin City–South alum addition to recent literature reviews of applications to lakes to precipitate or inactivate phosphorus.

Source	Dose	Notes
Volumetric (mg Al/L):		
Cooke & Kennedy (1981)	Mean: 9.6± 7.4 Median: 7.7 Range: 0.5–26 m	27 lake applications
Twin City–South: this study	Whole lake: 0.81 Epilimnion: 1.45	Calculated from dose assuming uniform mixing from (0–20 m)
Areal (g Al/m^2):		
Smeltzer (1990)	Mean: 43 ± 37 Median: 32 Range: 18–139	8 hypolimnetic and metalimnetic applications
Twin City–South: this study	19	46% of water column total P (353 kg) removed July–September 1990

Ultimately, the restoration plan was never formally approved by the MPCA, and the alum application was rejected on the basis of potential risks to public health and the environment of aluminum in natural water bodies (Axler et al. 1996). The rapid recovery of the lake (and the subsequent bankruptcy of MAI) made it unnecessary for MAI to contest these rejections. A later application to add ferric chloride to the Sherman pit lake during aquacultural operations as a means of complying with their in-lake P standard was also denied, pending further review of the state's policy regarding the application of precipitants to any "waters of the State" (MAI 1995). The 1990 alum treatment created a local controversy over the public health aspects of adding aluminum to potential drinking water sources, and although we believe this was unjustified (Axler et al. 1992*b*, 1996), the general issue remains a potential impediment to the use of metal salts for lake restoration in Minnesota.

Sealing of Lake Bottoms

Although the addition of aluminum salts should greatly reduce the amount of internal P regeneration from the sediments, it would not be expected to significantly affect the sediment oxygen demand associated with aquacultural organic wastes. Therefore, to maintain an aerobic hypolimnion for minimizing P release, either the water column must be aerated or the organic wastes on the bottom capped. Indefinite intensive aeration, although feasible, was felt to be unnecessary because the coring studies suggested that burial by low-organic-matter, low-P soils from basin erosion would effectively mitigate both O_2 consumption and P release from aquacultural sediments by increasing the diffusive barrier between wastes and the overlying water.

To further demonstrate the feasibility of sediment "capping," a series of microcosm studies was conducted with TC-S sediment and basin mineral soils to simulate potential whole-lake treatments. Combinations of capping with and without concurrent alum application showed that capping the sediment could effectively reduce the sediment oxygen demand (SOD) and decrease the release of phosphorus, particularly if both treatments were made together (Table 4). These data showed that capping the sediments with 0.6 –10 cm of mineral soils in concert with a 25–50–g/m^3 application of alum would effectively reduce P release during anoxia (56–99%), and a cap of 0.6–10–cm of sand or basin soils would reduce the rate of sediment oxygen demand by 24–82%. Oxygen depletion as a function of the thickness of the sediment "cap" (t, in cm) was well described by the equation

$$\text{SOD}\ (\text{mgO}_2/\text{m}^2/\text{day}) = 775 - (655) \cdot (t/[t + 0.965]),$$

Alum treatment alone prevented P release and also removed additional phosphorus from the overlying water (Table 4).

The volume of material necessary to provide a cap of up to 5 cm over the entire lake bottom would be enormous, 10^5 m^3, so the most feasible source of material was the basin walls themselves. The method proposed was a process called "hydraulicking," in which a high-velocity jet of water would be directed against the basin walls to accelerate the natural erosion already occurring (see Figs. 3, 11). The restoration plan estimated that about four months of hydraulicking at 3800 gpm from a 3-inch nozzle at 200 pounds per square inch would be required to deposit a layer of 20 cm of material on the bottom, which was the "worst-case scenario." Higher amounts of fines would probably be deposited in the mid-lake basin, and continuing the intensive aeration and circulation used previously would help make the layering more uniform (MAI 1994).

Although the capping proposal would simply have accelerated the rate of naturally occurring erosion of inorganic mineral soil into the lake—the estimated requirement was for about two years of "typical" erosional load based on the Natural Resources Research Institute (pre-1993) core data—the proposal was rejected by the MPCA because of the potential policy implications associated with endorsing the acceleration of shoreline erosion in a water body and violating the state's turbidity standard of 25 NTU (even if only for a short time). A permit was required from the U.S. Army Corps of Engineers, Clean Water Act Section 404, as well as a permit from the MPCA, Clean Water Act Section 401, and a water permit from the Minnesota Department of Natural Resources because of the proposed "shoreline" modification. Ironically, the February 1994 freeze core survey (Table 3) indicated that aquacultural wastes had already been buried by at least 3.7 cm of mineral soil, and the water quality data showed that dramatic improvement had already occurred after only one growing season of fallowing. The alum application required an additional environmental assessment worksheet from the state, and the permit was ultimately denied. In November 1994 MAI declared bankruptcy and the "active" sediment sealing components of the restoration plan became moot.

Alternative Regulatory Options

We believe that nutrient concentrations in the aquaculture pit lakes could have been more effectively reduced, for a given rate of fish feeding, by allowing flexibility in rates of aeration and vertical mixing during the summer. Stringent water quality standards prohibiting hypolimnetic anoxia effectively prevented significant amounts of nitrate and nitrite removal by bacterial denitrification, an obligately anoxic process, from occurring in bottom waters and possibly surficial sediments. During the summers of 1990 and 1991, loss rates in TC-S and Sherman pit lake during periods of deep hypolimnion anoxia ranged from 52 to 374 mg N/m^2/day based on mass balance changes in DIN in deepwater strata. These areal rates were normalized to the approximate area of the stratum where low oxygen conditions began (<1 ppm O_2). The values were high but not atypical of rates determined in other aquatic systems where denitrification is not limited by available substrate (Seitzinger 1988). Over a 90-day period of anoxia for water less than 45 m deep at TC-S and less than 50 m at Sherman pit lake, we estimate that 0.035–0.269 mg N/L could have been removed on a whole-lake basis. This amount would have been a potentially significant fraction of the average annual DIN increment measured at TC-S from 1990 to 1993 ($\sim$0.6 mg N/L) and Sherman pit lake from 1991 to 1994 ($\sim$0.8 mg N/L). Unlike other potential removal processes, it represents a total removal because the end product of the process is predominantly N_2 gas, which is not an environmental or public health concern.

A management option of providing some control of nitrate buildup by denitrification would require regulatory flexibility in that any requirement for maintaining strictly aerobic conditions in bottom water would have to be relaxed for some period of time in the summer growing season. Although P release from anoxic sediments would probably increase, it would not be likely to affect water quality because increased P levels would occur far below the euphotic zone (which typically extends to 10–20 m at most) and far below the surface water where P standards apply. This bottom water could be aerated again in the fall to assist natural lake turnover in re-precipitating released phosphate and resaturating the water column with oxygen, both of which are also in the best interests of an aquacultural enterprise. Attention would have to be paid to hypolimnetic nitrate concentrations because anoxia will only remove nitrates and nitrites, not ammonium. A strategy involving alternate periods of aeration to promote nitrification of ammonium to nitrate, followed by stagnation to allow for denitrification, could be implemented.

Periods of reduced midsummer aeration would enhance the sedimentation (and net rate of burial) of nitrogen and phosphorus. Reduced aeration would lead to short-term (i.e., growing season) increases in algal production by allowing the lake to restratify to a depth of about 5 m, the "natural" midsummer thermocline depth (Table 1). Thus, phytoplankton in this narrower mixed layer would be exposed to much higher average light intensity than in previous years when the lake was artificially mixed. The associated increase in DIN and ortho-P assimilation into particulates (algae) would increase the net sedimentation loss from the water column. Reduced vertical circulation, relative to that used in TC-S in 1993 and at Sherman pit lake from 1991 to 1994, would also reduce the resuspension of settling solids and surficial sediments. Reintroduction of these materials into more oxygenated water is undesirable in the restoration context because this would increase the mineralization and solubilization rates of organic nutrients and therefore decrease the net loss of nutrients to the sediments. Feed tests performed by MAI (unpublished reports) previously indicated that as much as 10% of the feed was fine particulates that were wasted in terms of fish growth but, more importantly, settled slowly and were resuspended more easily, contributing to water column oxygen depletion and nutrient pools (Axler et al. 1993).

The importance of resuspension due to intense mixing is suggested by the continued relatively high concentration of phosphorus ($>$200 ug/L for most of 1994 [MAI 1994; Axler et al. 1996]) in Sherman, despite similar volumetrically weighted rates of feeding to TC-S (Axler et al. 1996). The lake volume–normalized annual rates of fish food added to Sherman pit lake since 1993 were similar to the rates for TC-S during its peak production from 1989 to 1992. But the water column P concentration in 1993 and 1994 was more than twice the highest for values TC-S (Axler et al. 1996). Aeration intensity was increased lakewide at Sherman in 1993 to comply with the hypolimnetic dissolved oxygen standard in the 1992 stipulation agreement. The timing suggests that the artificial mixing resuspended solids that previously sedimented. Nitrogen levels were presumably less affected than P levels because most of the N load is from excreted ammonium, which is soluble and released from the net-pens directly into the upper water column. Consequently, changes in the solubilization of particulates due to higher dissolved oxygen and a longer residence time in the water column cause a proportionately smaller effect on the overall N pool than on the P pool.

TC-S was intensively mixed throughout most of the 1993 growing season in order to light-limit phytoplankton growth and prevent hypolimnetic anoxia. Although this may have worked in that chlorophyll *a* concentrations remained low (Fig. 5), it is not clear that it was needed for the environmental protection of the pit lake. No indigenous fish species were present to protect. In fact, the aquaculture business always had a vested interest in *not* allowing dissolved oxygen to decrease too much, because a severe buildup of reducing conditions (i.e., low O_2, high NH_3, high H_2S) would present a clear hazard to the health of their fish. Of all fish species, salmonids are among the most sensitive to oxygen and un-ionized ammonia.

Conclusions

Although society seeks economic means of reclaiming landscapes that have been degraded by mining, considerable opportunity exists for conflicts over alternative land uses and management of the new lakes created after the open pits fill with water. Water quality issues may arise from degradation associated with the geochemistry of the newly inundated lake basin, such as in precious metal mine pits (Miller at al. 1996), which may limit future uses of the lake and its "watershed." Where water quality is "good," the development of the resource, a primary reclamation goal, may be hindered or prevented if treated by regulatory agencies in the same manner as pristine or natural water bodies (Axler et al. 1992*b*, 1996).

Attention should also be paid to the important hydrologic differences between mine pit lakes, natural lakes, and even reservoirs. Pit lakes are characterized by long retention times, no surface discharge during decades of filling, and high erosional inputs of largely mineral soils. Their morphometry and mineral content may also predispose them to incomplete annual vertical mixing (i.e. meromixis). Although the application of standard water quality models may be problematic because model assumptions are violated (Miller et al. 1996; Yokom et al. 1997), particulate or particle-associated pollutants may in fact be rapidly and virtually permanently sedimented without human intervention. This in fact was demonstrated for phosphorus and organic matter at Twin City-South. In contrast, dissolved contaminants, such as nitrate, may be more persistent due to low hydraulic flushing and low biological productivity. Environmentally and economically efficient management of these significant new water resources will require careful attention to their unique limnological features, in addition to prudent consideration of the costs and benefits to the public and private sectors of their development and reclamation.

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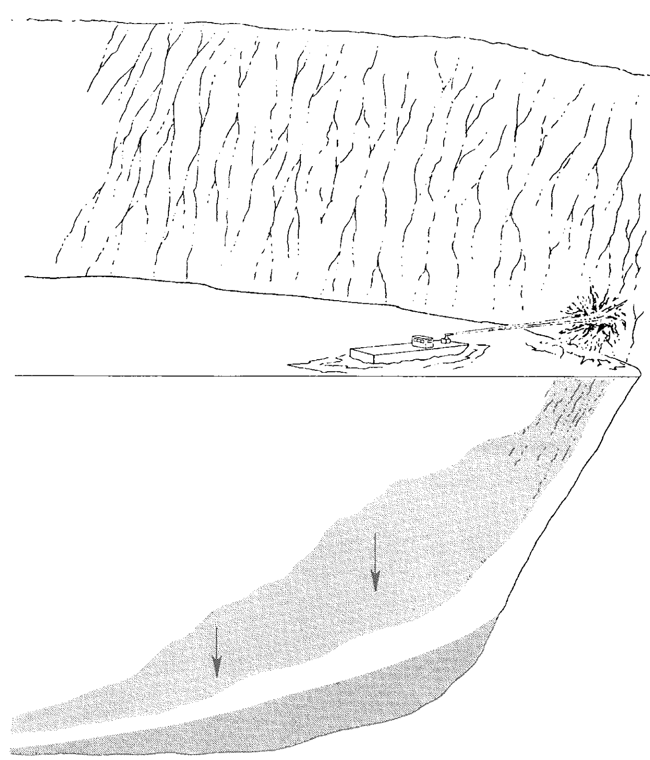

Figure 11. Schematic of hydraulicking.

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Appendix 1. Potential lake restoration techniques.

1. Source Controls
   - A. Treatment of inflows
   - B. Diversion of inflows
   - C. Watershed management (land uses, practices, NPS mitigations [BMP's])
   - D. Regulation or modification of riparian zones
   - E. Regulation or modification of industrial/business product

II. In-Lake Controls
   - A. Dredging
   - B. Volume changes (non-dredge, non-compaction)
   - C. Nutrient inactivation/precipitation
   - D. Dilution/flushing
   - E. Flow adjustment
   - F. Sediment exposure and dessication
   - G. Lake bottom sealing
   - H. In-lake sediment leaching (e.g. clay P-adsorption)
   - I. Shoreline modification and habitat manipulation
   - J. Riparian/wetland treatment of lakewater
   - K. Selective discharge
   - L. Mixing and/or aeration or oxygenation
   - M. Biomanipulation (food web alteration)
   - N. Algicides, herbicides, piscicides