

CASE 7.2

Clearwater Lake: Study of an Acidified Lake Ecosystem

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7.2.1 BACKGROUND AND RATIONALE

The lake and drainage basin which this study addresses is a small, oligotrophic water body, currently acidic and metal polluted. Since 1973, the lake has been part of the Sudbury Environmental Study by the Ontario Ministry of the Environment. The work of scientists in the MOE group provides a rather comprehensive data base of physical, chemical and biological information. It is

assumed that the present condition of the lake ecosystem has resulted from atmospheric inputs of H^+ , Cu^{2+} , and Ni^{2+} , both as a direct response to the pollutants as well as higher order effects. Experimental chemical manipulations have been made on nearby lakes, providing some information on the potential for reversibility of the effects of metals and acid.

Lake sediment chemistry combined with dating techniques is available to provide information on historical loading of substances to the lake. A study of diatom frustules in recent sediment provides some evidence for changes in species composition and biomass of diatoms over the past 200 years, which more than covers the period since smelting began in the early 1900's.

Over several years of study on calibrated watersheds, chemical and hydrological data on all inputs to and outputs from the lake and watersheds and mass balances were measured for the lake.

An opportunity therefore exists to present this as an ecotoxicological study at the ecosystem level, indeed, as one of the most complete of its kind. At the same time it has to be recognized that the objectives of certain study components were distinct from those of the present report.

7.2.2 TYPE OF ECOSYSTEM

Clearwater is an oligotrophic lake situated 13 km south-southwest of the INCO smelter near Sudbury, Ontario, on precambrian bedrock. Table 7.2.1 summarizes the physical characteristics. There are several cottages on the shoreline, and there is some recreational exploitation of the area. The major human impact has resulted from deposition of airborne pollutants on the lake and its watershed, and the major source of this pollution is assumed to be the Sudbury smelting complex. The chemical characteristics are presented in a later section, since the chemistry has been extremely modified by the inputs of pollutants.

Table 7.2.1 Clearwater Lake: physical characteristics.
(Sources: Dillon *et al.*, 1979; Scheider and Dillon, 1976;
MOE, 1981)

Lake area A_0 (ha)	76.5	Bedrock geology: gneiss, migmatite, quartzite
Lake volume $V(10^5 m^3)$	64.2	Shoreline length L (km) 4.97
Drainage area A_d (ha)	34.0	Shoreline development 1D_L 1.6
Mean depth \bar{Z} (m)	8.3	Flushing rate $\rho(y^{-1})$ 0.4
Maximum depth Z_m (m)	21.5	Coordinates 46° 22' latitude 81° 3' longitude

$$^1D_L = \frac{L}{2\sqrt{\pi A_0}}$$

7.2.3 POLLUTANT INPUT

The major toxic pollutants are H^+ , Cu^{2+} , and Ni^{2+} . For any air pollution scenario, impacts of a local source have to be superimposed upon the regional patterns of air pollutants resulting from long-range transport. For the current status of local versus regional contributions of air pollutants to Clearwater Lake, information is available from a recent study made during an 8-month shutdown of the INCO operation (Scheider *et al.*, 1981). Bulk deposition of hydrogen ions, sulphate and a number of metals was compared prior to and during the shutdown which occurred from September 1978 to June 1979. The results indicated that total copper, nickel and sulphate, over a 12 km radius, decreased by more than 90 per cent during shutdown, supporting the hypothesis that smelting and related operations in the Sudbury area were contributing significantly to the materials deposited on the lakes in the Sudbury area. In contrast, hydrogen ion deposition did not decrease significantly during the same period. The interpretation of this statistic is discussed (MOE, 1980) and comparisons made with deposition measurements for Muskoka-Haliburton, 225 km away. In this latter region, H^+ deposition *increased* during the Sudbury shutdown period, suggesting that the apparent lack of H^+ decrease in Sudbury during shutdown was a result of increased input from long-range transport. This assumes that Sudbury and Muskoka-Haliburton are affected by the same pattern of long range transport, from the same regional air masses. If this assumption is correct, and if there were no local sources of H^+ , then the Sudbury region should have shown a comparable increase over the same period. Since there was no such increase, the authors concluded that there was in fact a local *decrease* in H^+ during the shutdown period; from this it follows that there is normally a local source of acid deposition (Dillon, 1981).

Table 7.2.2 Annual bulk deposition of metals, nitrate and sulphates in Sudbury and Muskoka-Haliburton areas. (Sources: NRCC, 1981; Jeffries and Snyder, 1981)

Element/ion	Annual average deposition ($mg\ m^{-2}\ yr^{-1}$)	
	Sudbury S ¹	South-Central Ontario ²
NO_3^-N	553	350–470
SO_4^{2-}	5300	3000
Cu	136	2.12
Ni	84.0	1.6
Zn	68.7	15.7
Al	53.9	58.8
Mn	5.79	7.56
Fe	2.12	84.7
pH	3.8–4.5	4.03–4.38

¹January 1976–June 1978

²October 1977–September 1978

Information of the type discussed above emphasizes the fact that the contemporary situation for pollution of Clearwater Lake is complex, resulting as it does from a combination of local and regional air pollution. The 'experiment' of the 8-month shutdown permits some clarification of this. However, the authors point out that it does not clarify the situation concerning historical causes of acidification of lakes and watersheds in the Sudbury area.

Rates of deposition for sulphate and metals in precipitation in the Clearwater Lake area are given in Table 7.2.2. These values are for bulk (i.e. wet and dry) deposition. Values for Muskoka-Haliburton (South-Central Ontario) are provided for comparison. The inputs of copper, nickel and sulphate for the Sudbury location are extremely high. Mass balance studies for the calibrated watershed indicated that SO_4^{2-} output was substantially greater than the input measured as bulk deposition, leading to the tentative conclusion that an additional input directly onto the lake surface was occurring. The most reasonable explanation with the current state of knowledge is that there are direct inputs from SO_2 in the Sudbury area (Dillon, 1981). In support of this, the fraction of SO_4^{2-} unaccounted for in the mass balance decreased with distance from Sudbury.

Table 7.2.2 indicates that the pH of the bulk precipitation was low, but no lower than that of precipitation in other parts of Ontario including the South-Central region.

In summary, the major inputs of pollutants to the lake are:

- Hydrogen ions: local source and long-range transport distinguished by comparisons between operational and shutdown periods
- Sulphate ions: predominantly local source, but some long range transport
- SO_2 : local source, assumed from the mass balance studies
- Cu^{2+} and Ni^{2+} : local source

Each of these components currently impinge directly onto the lake surface, and most of them also come from the watershed. A large pulse of metals and acid occurs with the spring run off. In any terrestrial ecosystem, the chemical composition of rainfall is modified by the tree canopy and by interaction with soils. The watershed of Clearwater Lake is itself disturbed and lacks the forest which is otherwise typical of this region. Without the tree canopy, and with soils themselves polluted, the chemistry of water entering the lake is different from rainwater. In particular, calcium and magnesium are leached from soils by acidic water, and Clearwater Lake has higher levels of these cations than the more remote lakes (Table 7.2.3).

The major nutrients, phosphorus and nitrogen, are of considerable significance to the biota in lakes. The levels of total phosphorus in Clearwater Lake in the 1970s varied from 2.4 to 5.9 $\mu\text{g l}^{-1}$ (MOE, 1981), which is not atypical of soft-water Precambrian Shield lakes (Armstrong and Schindler, 1971). Total nitrogen to total phosphorus ratios ranged from 25:1 to 80:1, indicating that P was in shortest supply (Schindler, 1977).

Table 7.2.3 Clearwater Lake, major ions 1973–79, whole lake composite samples. (Source: Dillon *et al.*, 1979)

	1973	1975	1977	1979	ELA ²	M-H ³
conductivity, $\mu\text{mhos cm}^{-1}$, 25°C	89	85	89	85	19	33
H^+	51	47	79	39	1	1
\sum cations (H^+ , Ca^{2+} , Mg^{2+} , Na^+ , K^+)	616	504	567	561	203	271
SO_4^{2-}	579	519	550	458	63	170
HCO_3^-	0	0	0	0	62	110
\sum anions (SO_4^{2-} , Cl^- , $\text{HNO}_3^- - \text{N}$, HCO_3^-)	—	566	616	565	164	298

¹ All ions in $\mu\text{eq l}^{-1}$.² Values for mean of 40 lakes in the Experimental Lakes Area (Armstrong and Schindler, 1971).³ Muskoka-Haliburton, South-Central Ontario.

Inputs of nitrate from the atmosphere in precipitation comprise a significant component of the nitrogen budget in any ecosystem, and normally almost 100 per cent of the incoming nitrate is retained in a vegetated watershed. Levels of nitrate were high in Clearwater Lake, as they are in most severely acidified lakes in the area most affected by the Sudbury operation. The major reason for this is believed to be the lack of vegetation in the disturbed watersheds and associated losses of nitrate to the lake (MOE, 1981).

7.2.4 POLLUTANT BEHAVIOUR

For an aquatic ecosystem, the chemical composition of the water and sediments at any point in time is related to the past history of inputs, outputs and transformations of chemical pollutants, as well as the 'natural' geochemistry of the region. Since there is a flushing of water in a lake, water chemistry reflects inputs and outputs of materials for a lake over a short time scale of years or even months. Tables 7.2.3 and 7.2.4 show the major ions and some of the metal ions measured in Clearwater Lake over the past 8 years. Values for other areas are included for comparison. As might be anticipated from the chemistry of precipitation, H^+ , sulphate and metal ions in the water are exceptionally high. Bicarbonate is absent and is replaced by sulphate as the major anion, a feature which is characteristic of lakes acidified by acid sulphate deposition. High aluminum concentrations, typically associated with acidification, result from dissolution of Al from soils and bedrock, i.e. aluminum can be perceived as an internally generated contaminant.

The form or species of a metal in water has an effect on its toxicity to biota, mainly by determining its availability. In general, as conditions become more acidic, more of the metal exists in the 'free' or ionic form which, for most

Table 7.2.4 Clearwater Lake, total metal concentrations 1973–79, whole lake composite samples. (Source: Dillon *et al.*, 1979)

	1973 ²	1975	1977	1979	ELA ¹	M-H ⁴
³ Copper	98.3	100	81.3	59.8		< 2
Nickel	283	272	278	220		< 2
Zinc	47.9	49.0	39.2	31.4		5
Iron	114	132	88.2	55.0		98
Aluminum	420	447	381	272		42
Manganese	328	289	290	282		33
pH	4.3	4.3	4.2	4.4	5.6–6.7	5.6–6.4

¹Experimental Lakes Area, Patalas, 1971.

²Sample was 1 m from bottom in 1973.

³All in mg m⁻³ (ppb).

⁴Muskoka-Haliburton, South-Central Ontario.

elements, is the form toxic to biota. Analysis of copper and zinc by anodic stripping voltammetry (Chau and Lum-Sheu Chan, 1974) indicated that 50–80 per cent of these metals in Clearwater Lake were in a free or weakly bound form (MOE, 1981).

In contrast to water chemistry, which reflects only recent years or decades of inputs and outputs, lake sediment chemistry can often be used to provide historical information on a lake and its drainage area over its entire life span.

Sediment cores collected from Clearwater Lake were dated by ²¹⁰Pb. Sedimentation rates increased at a point about 50 years (± 5) before present, but there are uncertainties about recent years because of a mixing of the top 3 cm of the sediment profile. Nevertheless, there is evidence from the sedimentation rates to support the idea that disturbances in the land have occurred over the last 50 years, which coincides with changes in vegetation cover resulting from smelter fume and related damage.

Metal concentrations in recently deposited sediments of Clearwater Lake are shown in Table 7.2.5, with ratios of peak to background levels. The values indicate surface enrichment of Cu, Ni, Zn, Pb, Cd and Fe, and a decrease of Al. This decrease in Al results from acid leaching of Al from sediments.

Profiles of metal concentrations provide a chronology for the deposition of metals, and it is concluded that:

1. The elevation of Cu, Ni, Zn, Fe and Cd, and the decrease in Al occurred 50 years (± 5) before present.
2. The elevation of Pb began considerably earlier than 50 years ago, but continued to increase so that the highest levels of Pb are found in the last 3–4 decades (MOE, 1981).

Sediments are generally regarded as sinks for metals. In a series of experimental aquaria, release of copper and nickel across the acidic sediment water interface

Table 7.2.5 Clearwater Lake, metals in sediments. Peak metal concentrations in recently deposited sediments, background levels and ratio of peak to background concentrations for Clearwater Lake sediment cores. Bracketed Al concentrations correspond to recent low values. (Source: MOE, 1981)

	Cu $\mu\text{g g}^{-1}$	Ni $\mu\text{g g}^{-1}$	Zn $\mu\text{g g}^{-1}$	Pb $\mu\text{g g}^{-1}$	Cd $\mu\text{g g}^{-1}$	Al mg g^{-1}	Fe mg g^{-1}
Core #1 Peak	630	1300	96	89	0.90	(12)	83
Bkgd	37	56	59	4	0.28	27	30
Ratio	17	23	1.6	22	3.2	—	3
#2 Peak	1700	1600	350	150	6.70	(20)	91
Bkgd	63	76	145	3	0.96	28	32
Ratio	27	21	2.4	50	7.0	—	3
#3 Peak	1500	1600	290	160	5.10	(19)	79
Bkgd	60	60	143	3	0.82	25	27
Ratio	25	27	2.0	53	6.2	—	3
#4 Peak	2000	1800	390	150	6.20	(15)	34
Bkgd	83	40	93	3	0.63	21	14
Ratio	24	45	4.2	50	10	—	2
#5 Peak	620	1300	86	120	0.60	(14)	110
Bkgd	38	40	108	4	0.73	24	21
Ratio	16	33	<1.0	30	<1.0	—	5
Average ratio	22	30	2.6	41	6.6	—	3

and into the overlying water was demonstrated for Clearwater Lake sediments which were oxygenated and contained living organisms (Stokes and Szokalo, 1977). If this process is important in the field, the long-term fate of the lake could be affected, even in the event that inputs of metal were controlled or prevented.

An analysis of diatoms in sediment cores was made by Walsh (1977). Her study showed that surface sediments were of lower pH than deeper sediments, and that acid-tolerant diatoms predominated in the top 6–10 cm of each core. Diatoms of alkaline preference were more common below 10 cm and pH-indifferent species were present at all depths but were more common below 4 cm. Cores were dated by ^{210}Pb and by the *Ambrosia* rise, and sedimentation rates were estimated at 1 cm per 10 years. Thus the major shift towards acidic diatoms occurred over the last 60–100 years, which coincides with the development of smelting in the Sudbury area over the past 70 years. Other lakes in the area produced similar results, providing evidence linking acidification with the smelting activities.

By their ecological preferences, diatoms are believed to be reliable indicators of pH (Nygaard, 1956; Patrick *et al.*, 1968) but their preferences for metals are less well known and they may be less useful as indicators of historical metal contamination.

At this point it is appropriate to raise the problem of distinguishing between

the effects of acidity and metal contamination in a lake which has both. Clearwater is acidic and polluted by nickel and copper, and the chemical composition of the water is atypical (e.g. sulphate replaces bicarbonate). The effects of lake acidification *per se* are complex, involving direct effects of H^+ on biota, community interactions resulting from changes in community structure, release of metals or change in speciation of existing metals and the response of biota to the metals. Clearly, without experimental evidence, it may not be possible to attribute the biotic changes which have been observed to one specific toxic material. It will become apparent, however, that by comparison with other systems, rather specific interpretations can be made of some of the data in terms of cause and effect.

7.2.5 EFFECTS ON SPECIES AND COMMUNITIES

7.2.5.1 Phytoplankton

Clearwater Lake is extremely transparent and one would expect this clarity to have physical and biological effects. The epilimnion of Clearwater Lake was usually 10–12 m thick compared with the more normal 6–7 m in comparable sized nonacidic shield lakes (MOE, 1981). NRCC (1981) consider that the clarity of the acidic lakes is due to precipitation of organic matter by aluminum and not to reduced algal biomass. Schindler (1980) similarly found increased clarity but no decrease in chlorophyll in an artificially acidified lake. The total annual biomass of phytoplankton in Clearwater varied from 0.3 to 0.7 mg l⁻¹, which is not significantly lower than lakes in Muskoka-Haliburton (Yan, 1979).

Extinction coefficients for Clearwater, measured in 1978, were 0.13 microeinstains m⁻² s⁻¹ compared with 0.56 for Harp Lake and 0.41 for Red Chalk Lake, both oligotrophic lakes in South-Central Ontario (NRCC, 1981). Because of the unusual light regime, vertical distribution of phytoplankton may be altered in acid lakes, and increased cell concentration may occur at greater depths than in neutral lakes. In fact, vertical chlorophyll profiles indicated that phytoplankton did accumulate in the deeper strata of Clearwater Lake.

Biomass of phytoplankton was originally considered to be decreased in acid lakes. Hendrey *et al.* (1976) suggested this but other data do not consistently support their conclusions. The discrepancies may well be the result of technical inconsistencies. It is of great importance to determine biomass from a sufficient number of samples over the season and from sufficient depths. In Clearwater Lake, there was considerable seasonable variability in biomass. Very significant factors in sampling are the clarity of the water, the depth of the epilimnion and the vertical distribution of phytoplankton in acid lakes. Samples taken from the surface of the epilimnion are, therefore, unlikely to be representative of the phytoplankton community and this may have led to premature conclusions in early studies.

In comparison with softwater shield lakes in the ELA and Muskoka-Haliburton, the number of genera in the phytoplankton of Clearwater Lake was low, and the flora was deficient in the number of genera of Chrysophyceae, Cyanophyceae and Cryptophyceae (Table 7.2.6). The most common genera of algae in Clearwater Lake in a 1973–77 survey were *Peridinium* (Dinophyceae), *Cryptomonas* (Cryptophyceae), *Dinobryon* (Chrysophyceae), *Chlamydomonas* and *Oocystis* (Chlorophyta). Numbers of genera of diatoms and desmids were comparable to those in the less acid lakes, which is not surprising since acid tolerant diatoms and desmids are known to occur (Yan, 1979).

Metal tolerance of algae taxa is less well known, but Yan (1979) suggested that algae present in metal-contaminated water were metal tolerant, pointing out that for other lakes in the Sudbury area, metal tolerance of algae has been demonstrated (Stokes *et al.*, 1973). However, the presence of the algae in metal-polluted waters is only circumstantial evidence that they are metal tolerant, and in some instances the algae when tested under laboratory conditions were less tolerant than would have been anticipated from the total metal levels in the lake (Stokes *et al.*, 1973; Stokes, unpublished). Factors which determine effective metal concentrations include organic content and pH; in other words, not all of the measured metal is 'seen' by the algae. This would be one explanation why tolerance is lower than expected based on total levels. No metal tolerance tests have been made on algae isolated from Clearwater.

In summary, the phytoplankton biomass of Clearwater Lake was not significantly lower than that of neutral softwater lakes, although its vertical distribution was affected by acidification. The species composition and community structure, however, were markedly altered, with decrease in richness and a marked predominance (30–35 per cent of biomass) of dinoflagellates, especially *Peridinium inconspicuum*. This species has been described as dominating the

Table 7.2.6 Clearwater Lake, phytoplankton genera represented. (Source: Yan, 1979)

	Number of genera		
	Clearwater	ELA	Blue Chalk (M–H)
Chlorophyta (excluding Desmidiaceae)	17	32	25
Chrysophyceae (including Prymnesiophyceae)	10	22	15
Cyanophyta	6	18	11
Bacillariophyceae	12	7	11
Cryptophyceae	3	6	6
Desmidiaceae	7	5	4
Dinophyceae	4	5	3
Xanthophyceae	0	4	1
Euglenophyta	1	4	0
Total genera	60	103	76

phytoplankton biomass in a number of lakes which were acidic but not nickel and copper contaminated (e.g. Hörnström *et al.*, 1973; Yan and Stokes, 1976). This species is thus considered indicative of acid lakes and suggests that the low pH of Clearwater Lake may influence the phytoplankton composition more than do the metals.

Any phytoplankton which survives at low pH must have the capacity to use CO₂ as its carbon source. Furthermore, experimental work has shown that there is a change in the permeability to ions of the cell membrane at low pH (Mierle and Stokes, 1976; Mierle, personal communication) so that, depending upon the transport mechanisms for nutrient uptake, there may be profound effects on nutrition at low pH.

7.2.5.2 Zooplankton

In Clearwater Lake, zooplankton showed low species richness compared with less acid lakes—an average of 3.7 species of crustacea per sample compared with 11.9 for lakes in Muskoka-Haliburton—and a predominance of one species, *Bosminia longirostris* (Table 7.2.7). *Cyclops vernalis* was often codominant in early spring. *Daphnia* species, important in nonacid lakes, were completely absent from Clearwater. Although biomass of zooplankton in Clearwater was significantly lower than that of the nonacidic lakes, correlation analysis revealed no significant relationship between zooplankton biomass and pH, copper or nickel in a set of acidic lakes which included Clearwater (Yan and Strus, 1980). The lack of such relationships was unexpected, but may have been related to the rather narrow range of pH included in the data set. There were some similarities between the zooplankton species composition of Clearwater and other acidic non-metal-contaminated lakes but one notable difference was in the dominance of *Diaptomus minutus* in acidic lakes (Sprules, 1975; Haydu, personal communication) while *Diaptomus* was very infrequent in Clearwater Lake. Thus there is fairly good evidence that metals influenced the crustacean zooplankton species composition. This is supported by the fact that copper levels alone in

Table 7.2.7 Clearwater Lake, crustacean zooplankton. (Source: Yan and Strus, 1980.)

	Clearwater Lake 1976–78 (Sudbury)	Blue Chalk Lake 1977 (Muskoka-Haliburton)
Average no. species per collection	3.7	11.9
No. animals per litre	14–24	43
Biomass of <i>Bosminia longirostris</i> (% of crustacea)	79–93	0.3
Biomass of copepods (% of crustacea)	6–20	48.4

Clearwater exceeded the 16 per cent reproductive impairment concentrations for *Daphnia magna* reported by Beisinger and Christensen (1972).

Changes also occurred in the rotifer community. The rotifers contributed 16–30 per cent of the total zooplankton biomass in Clearwater, compared with approximately 1 per cent in South-Central Ontario lakes. Furthermore the dominant species *Keratella taurocephala* formed up to 98 per cent of the rotifer biomass in Clearwater Lake; this species was also dominant in the acidic lakes of the La Cloche Mountains (Roff and Kwiatkowski, 1977) although rotifers as a group were not abundant in the latter study.

7.2.5.3 Macrophytes

Softwater lakes have typically isoetid-dominated macrophyte flora. In Clearwater Lake *Ericaulon septangulare* was the dominant macrophyte. In this respect it resembled the nonacid lakes of Muskoka-Haliburton. Species richness was poor, with a total of 7 compared with 11 in the nonacid lakes (Table 7.2.8). Macrophyte growth extended to 8 m in Clearwater Lake, compared with 3–4 m in the nonacidic lakes. These values approximated the secchi disc readings (Wile *et al.*, 1981).

The average macrophyte biomass was high in Clearwater Lake (Table 7.2.8)

Table 7.2.8 Clearwater Lake, macrophytes. A: abundant, C: common, O: occasional, R: rare. (Source: Wile *et al.*, 1981)

Species	Clearwater Lake Sudbury	Red Chalk Lake Muskoka-Haliburton
Vascular plants		
<i>Eleocharis acicularis</i> R. & S.	O	R
<i>Eriocaulon septangulare</i> With.	A	A
<i>Isoetes</i> sp.	—	O
<i>Juncus pelocarpus</i> Mey.	O	O
<i>Lobelia dortmanna</i> L.	—	C
<i>Lycopus</i> sp.	R	—
<i>Myriophyllum tenellum</i> Bigel.	O	O
<i>Nuphar variegatum</i> Engelm.	—	O
<i>Sparganium</i> spp.	—	O
<i>Utricularia purpurea</i> Walt.	—	R
<i>Utricularia vulgaris</i> L.	—	R
Mosses and liverworts		
<i>Cladopodiella fluitans</i> (Nees) Buch.	O	O
<i>Drepanocladus exannulatus</i> (BSG) Warnst.	C	C
No. of macrophyte taxa	7	12
Average macrophyte biomass (g m ⁻² dry weight)	510 ± 98	71.6 ± 11.2

compared with the nonacidic lakes, whose macrophyte biomass was within the ranges reported in the literature. The high biomass in Clearwater Lake is indicative of the tolerance of the angiosperm *E. septangulare* and the bryophyte *Drepanocladus exannulatus* to low pH and elevated metals; biomass values are nevertheless surprisingly high (Wile *et al.*, 1981) and the authors have not proposed any other explanation.

The simplification which apparently occurs in communities of plants and animals during the acidification process is likely to change many interactions such as competition, grazing and predation, and, in general, it is not surprising to find an increase in the biomass of certain constituents of the food chain. No specific mechanism for the observed macrophyte biomass is suggested.

7.2.5.4 Bacteria

Very little data are available on the reducer communities in the lake. A comparison between the acidic Clearwater and comparable nonacidic lakes has been made by Thompson and Wilson (1973) who considered that the composition and population levels for yeasts and moulds in Clearwater was within the normal range for oligotrophic lakes. Ammonium oxidizing bacteria were not found, indicating limited nitrogen cycling.

Indirect evidence for decreased heterotrophic aerobes in Clearwater Lake comes from comparisons with the neutralized Lohi Lake in which the heterotrophic plate count increased significantly after liming. Concerning specific groups, the sulphate reducers in sediments were relatively low in Clearwater Lake, compared with eutrophic or neutralized oligotrophic lakes even though the necessary substrate was abundant in Clearwater (Scheider *et al.*, 1975). There were relatively low heterotrophic: aciduric ratios—0.176 (1 m below surface) and 0.276 (1 m above bottom) (Scheider and Dillon, 1976).

Functional studies of the heterotrophic microbiota are lacking.

7.2.5.5 Benthos

The benthic macroinvertebrate community in Clearwater Lake was dominated by chironomids; certain taxonomic groups, the molluscs and ephemeroptera, were absent. Actual numbers of benthic invertebrates ranged from 654 to 1172 m⁻², which is similar to the values in ELA lakes of 518 to 1642 m⁻² (MOE, 1981).

7.2.5.6 Fish

No fish were caught in extensive trapping in 1972, no fish were recorded in the lake during the study period 1973, and no reliable historical records are available. Local residents recall seeing fish in a number of the lakes around Sudbury, which

suggests that 40–50 years ago the lake did support fishery, but there was no consensus on the species present (Scheider *et al.*, 1975). Numerous studies have documented the disappearance of fish species with acidification (e.g. Beamish and Harvey, 1972) and it is not surprising that Clearwater at pH 4.3 had no fishery. However, the relative importance of metals and hydrogen ions and the time trends of the loss of fish remain unknown.

7.2.6 ECOSYSTEM FUNCTION

7.2.6.1 Shifts in Primary Production

Subjective observations have indicated that with lake acidification, primary production shifts from the pelagic to the littoral and benthic. Many acid lakes have prominent benthic mats of algae and mosses (e.g. Grahn *et al.*, 1974; Hendrey and Vertucci, 1980; Stokes, 1981), and this benthic production is apparently facilitated by the clarity of acid lakes. In Clearwater Lake there was a very high macrophyte production in the littoral zone with a resulting increase in the macrophyte: phytoplankton biomass ratio. To date, no prominent algal or moss mats have been recorded in Clearwater, although the nearby Swan Lake, whose chemistry is very similar, has dense mats of *Pleurodiscus* from the littoral to the mid-lake benthic region.

7.2.6.2 Productivity

Limited data on primary productivity for Clearwater Lake suggest, by comparison with the Muskoka-Haliburton lakes, that there was no significant decrease (Dillon *et al.*, 1979). However, the authors stress that differences in light regimes were not taken into account and that sample numbers were small. Using Schindler and Nighswander's (1970) production efficiency calculation, MOE (1981) showed that efficiencies of conversion of light energy to carbohydrate were similar for Clearwater Lake (Sudbury) and Blue Chalk Lake (Muskoka-Haliburton), but they caution that conclusions are tentative. Nevertheless, this finding would be consistent with the other limited data in the literature on acidic lake productivity (Schindler, 1980). It is at first surprising that the metal burden does not appear to affect productivity in Clearwater Lake, since copper is known to be a potent inhibitor of photosynthesis (Steemann Nielsen *et al.*, 1969). However, if the organisms are metal tolerant then this would be a reasonable explanation for the lack of effect on productivity.

7.2.6.3 Community Interactions

It has been shown (Section 7.2.5) that species composition and community structure of phytoplankton and zooplankton in Clearwater Lake are different

from those in circumneutral lakes, and three hypotheses have been made concerning their interactions (Yan and Strus, 1980).

1. Only 50 per cent of the phytoplankton biomass in Clearwater is available as food for the zooplankton.
2. Changes in the structure of the phytoplankton in Clearwater are independent of zooplankton grazing.
3. The contamination of Clearwater results in reduced efficiency of energy transfer from phytoplankton to herbivorous zooplankton.

The dinoflagellate, *Peridinium inconspicuum*, which formed on average 45 per cent of the phytoplankton biomass, is a large organism (average diameter 14 μm) and was never detected in the gut contents of *Bosminia longirostris*, the dominant zooplankton. The balance of the phytoplankton present in appreciable quantity were of a size range considered 'edible'. However, the fact that zooplankton biomass in Clearwater Lake was substantially reduced while phytoplankton biomass and renewal rates were not reduced, makes hypothesis 1 unlikely. The capacity of the zooplankton to exert pressure on the phytoplankton, is also likely to be negligible in comparison with nonacidified lakes, so hypothesis 2 is supported.

Some studies have been made of rates of filtering by zooplankton and values by Haney (1973) for *B. longirostris* were used by Yan and Strus (1980) to estimate filtering rates. They considered that herbivore grazing would probably exert little effect on the phytoplankton, and concluded that for phytoplankton, pH *per se* controlled the community composition in acid lakes. This is supported by other data including Yan and Stokes' (1978) study on experimentally acidified columns in soft water lakes which were not metal polluted.

There is some evidence that the energy transfer from primary to secondary trophic levels is affected by acidification (Smith and Frey, 1971). Filtering rates estimated in Clearwater Lake were low (Yan and Strus, 1980). Recent studies on lakes of a wider range of pH, from pH 7.0 to 4.5 (Haydu, personal communication) have not demonstrated pH related decreases in filtering rates. However, Haydu's lakes were not metal contaminated. With the extremely low pH and additional contamination by Ni and Cu in Clearwater, it is still likely that energy transfer is less efficient.

It has been suggested recently that the role of invertebrate predators may be a major determining factor for zooplankton in acid lakes (Eriksson *et al.*, 1980). Reference is made in Yan and Strus (1980) to the predator *Cyclops vernalis* which appeared to control the *B. longirostris* population over each season but the long-term control by predators is not discussed further. In some acid lakes, predatory cyclopoids are reduced or absent (Haydu, personal communication). This aspect has not been evaluated in detail for Clearwater Lake, but remains as one of potential importance for an understanding of community dynamics.

7.2.6.4 Reduction and Nutrient Cycling

As a general observation, accumulation of detritus and decreased heterotrophic bacterial decomposition is said to accompany acidification in lakes. This has the potential ultimately to affect nutrient cycling, which would have a feedback effect on all communities. As far as is known, Clearwater Lake is no more nutrient limited than other nonacidic softwater lakes even though P and inorganic carbon are very low. However, more work is required on the effect of H^+ , Ni^{2+} and Cu^{2+} on the availability of these nutrients, as well as the effects of the pollutants on bacterial decomposition.

7.2.7 ECOSYSTEM RECOVERY

The chemical status of Clearwater Lake at the present time means that pH is rather stable, being buffered by systems other than the bicarbonate system, and it is likely to remain acidic for a long period of time, even if inputs of H^+ are decreased. For the metals, data during the 8-month shutdown (MOE, 1981) suggest that decreased inputs may be reflected by lower concentrations in the water. However, these data have not yet been reviewed as part of a long-term trend, and until data are available for the years after 1979 such a review cannot be done. It is also important to consider the role which sediments may play in cycling of metals through the water column.

Speculation can be made on the prospects for recovery resulting from intervention in the form of chemical treatments, since several lakes in the vicinity of Clearwater have been the subject of experimental manipulation. 'Recovery' implies a return to the original state and it is perhaps more accurate to use the term 'rehabilitation'. The major objective from a practical standpoint of manipulation of these lakes is to produce a viable fishery. From the scientific standpoint, however, well-designed experimental manipulations can also assist in elucidating the responses of biota to chemical perturbation.

7.2.7.1 Effect of Base Additions

In 1973, Lohi Lake, which is downstream of Clearwater, was neutralized by additions of $Ca(OH)_2$ (Dillon *et al.*, 1979). The pH rose to 7.0, but decreased rapidly and further additions of $Ca(OH)_2$ were made in 1974. Metal levels were substantially reduced by the base additions. Other chemical data are summarized in Table 7.2.9. There were no changes in total P and no hypolimnetic oxygen deficit as a result of the treatment.

After liming, secchi disc transparency decreased, bacteria increased and the biomass of phytoplankton decreased but recovered within a few months. Species composition of phytoplankton changed with a decrease in dinoflagellates and cryptophytes and an increase in chrysophytes, i.e. there appeared to be a reversal

Table 7.2.9 Lohi Lake, effects of base additions.
(Source: MOE, 1981)

Parameter	1973	1974
pH	4.4	6.1
Alkalinity $\text{mg l}^{-1} \text{ CaCO}_3$	<0	0.7
$\text{Ca}^{2+} \text{ mg l}^{-1}$	6.2	7.9
Total Cu mg m^{-3}	84	44
Total Ni mg m^{-3}	254	195
Total Zn mg m^{-3}	48	47

of the acidification process in the phytoplankton response. Zooplankton standing stocks were reduced and did not recover to resemble the communities of nonacidified lakes. The zooplankton biomass eventually recovered after 3 years, but a typical non-acidic community was not established. This is explained by the fact that the need for recruitment of certain zooplankton species may be limiting the process of recovery (Dillon *et al.*, 1979).

7.2.7.2 Effects of Phosphorus Additions

Lohi Lake was not fertilized but data are available for Middle Lake, another of the Sudbury study set. Phosphorus was added to raise total P from 2–5, up to $8 \mu\text{g l}^{-1}$ in 1975 and $12 \mu\text{g l}^{-1}$ in 1976. This lake had already been neutralized. Significant increases in phytoplankton biomass and impressive changes in species composition resulted from the phosphorus additions. Most notable was the increase in the biomass of blue-green algae (Cyanophyta), particularly *Mastigocladus*.

The phosphorus had less effect on the zooplankton; Middle Lake resembled Lohi in its zooplankton biomass as well as species composition. Although the increased phytoplankton biomass might lead to the expectation of an increase in zooplankton, the authors (Dillon *et al.*, 1979) point out that *Mastigocladus* is not likely to be a suitable food source for zooplankton.

7.2.7.3 Fish Stocking

As part of the liming study, fish were introduced into the study lakes in Sudbury (Gunn, personal communication; MOE, 1981). No fish had been caught in Clearwater, Lohi or Middle Lake in a 1972 survey. In 1976, after liming, Middle Lake had a pH of 6.2 and metal levels were 60, 470 and $35 \mu\text{g l}^{-1}$ for copper, nickel and zinc, respectively. Smallmouth bass (*Micropterus dolomieu*), Iowa darters (*Etheostoma exile*) and brook stickleback (*Culae inconstans*) were introduced and in 1977 no fish were captured by traps, gillnets or trap nets. In 1977, brook trout (*Salvelinus fontinalis*) were introduced into Lohi Lake, and

Table 7.2.10 Fish mortality in neutralized Sudbury lakes. (Source: MOE, 1981)

Lake	<i>n</i>	pH	Cu	Ni (mg m ⁻³)	Zn	GMST (days)
Middle	5	6.4	53	406	27	3.6(2.1–5.7) ²
Lohi	3	5.3	43	227	31	2.3(1.3–3.5) ³
Panache	1 ¹	6.5	6	28	6	34.0

¹ In this experiment fish were placed in transportation enclosures and driven over local roads for 2 hours prior to being placed into enclosures. In a later experiment fish were taken from the holding pool on the shores of Lake Panache and placed directly into the enclosure. No mortalities were observed for 50 days in this experiment, indicating that fish survival in the enclosures was possible given suitable water quality.

² Range for 5 experiments in Middle Lake.

³ Range for 3 experiments in Lohi Lake.

again no fish were recovered. The cause of death was concluded to be copper toxicity (Powell, 1977).

In order to test this assumption, experimental enclosures containing rainbow trout were introduced into Lohi and Middle lakes and also into Panache Lake to act as a control for the handling techniques. Table 7.2.10 shows the results. It was concluded that since low pH was not sufficient to have caused the observed mortalities, then metal toxicity was the cause of death. Metal concentration in the gills and liver of the Lohi and Middle Lake fish were higher than those in Panache. Although the liming had decreased the metal concentrations, the combined metal levels were still high enough to have accounted for the death of the fish (MOE, 1981).

7.2.8 Stability and Resilience of the Clearwater Lake Ecosystems

The concept of ecosystem stability is not far removed from the physicist's concept of stability: resistance to perturbation. The complexity of the ecosystem and the time scales of fluctuation may limit the usefulness of physical models in ecology, but conceptually such models have great potential to direct and organize the collection of ecological data, as well as to aid in their interpretation. According to ecological theory, communities and ecosystems have mechanisms which maintain stability. In this sense, stability refers to both structure and function. Components which are believed to affect stability include species diversity, spatial heterogeneity and population interactions such as competition, herbivore–food and predator–prey interactions. Not all ecologists agree on the details of the respective roles of these components and their interactions, but there is little dispute as to the importance of homeostatic mechanisms in the

maintenance of an ecosystem's viability. The stable state of a system most certainly is not a static state. Change is normal, which is why the ecosystem is often referred to as a dynamic system.

Changes result in variations of ecosystem properties such as population sizes and species composition which, when they occur in a stable or steady state ecosystem, fluctuate or oscillate around an average or 'normal' condition. Stability is also used in the context of the implicit or explicit assumption that response to a disturbance (a perturbation) is temporary, i.e. that upon removal of the disturbing factor the system will revert to the 'normal' condition. Holling (1978) has referred to this as 'a view of an infinitely forgiving Mother Nature'. This is not the only model of Mother Nature, however. Holling also illustrates by models of population responses to disturbance that there are alternatives to the above mentioned 'Beneficient Nature'. These include 'Ephemeral Nature', in which a disturbance results in instability and the fate is extinction of populations, and 'Mischievous Nature', in which a new domain of stability is eventually reached. For a more detailed discussion of the theoretical and practical aspects of these models, the reader is referred to Holling (1973, 1978).

Resilience is persistence under stress and often is a result of stress. To illustrate this by a simple example we can consider intertidal communities which are regularly exposed and covered by tidal movements, and contrast these with deepwater communities whose environment is much more stable in terms of water level. The regularly stressed intertidal communities are more able to tolerate or adapt to stress. In Holling's words, 'the continual "testing" of these systems gives them the resilience they have'. He juxtaposes the concepts of resilience and stability and considers that the resilience of a system is a more important characteristic of its viability than any fluctuation in the numbers of its components. In fact, 'protection' from influences which cause fluctuation may ultimately *decrease* resilience. Put very simply, the resilient properties of the components and structure of an ecosystem allow the system to persist.

In the case of Clearwater Lake, three questions are of interest concerning the state of the ecosystem:

1. Has the disturbance (60–80 years of metal and H^+ inputs) been so great that the system will not revert to its original state?
2. Is the present state a point on a trajectory towards extinction of whole functional groups, and continuing degradation of the system?
3. Is the system at, or directed towards, a 'new' stable state which is different from the original?

The greatest challenge still lies in knowing what to measure in order to answer these questions. No one has yet produced a satisfactory methodology to monitor an intact ecosystem, even though many pay lip service to the ecosystem or holistic approach. In a practical sense, we are still reduced to monitoring or experimenting with a small component of the system; given the 'right' set of parameters, this can

yield some very respectable results as the preceding chapters of this book have discussed. The alternative strategy is to 'measure everything' which is certainly unnecessary, but at present there is rarely an adequate theoretical basis for selection of the key or 'rate limiting' components.

In a consideration of the three questions posed above, information has been utilized from the large scale data gathering approach as well as from specific so-called 'reductionist' approaches.

With regard to the first question, which addresses reversibility of the change,

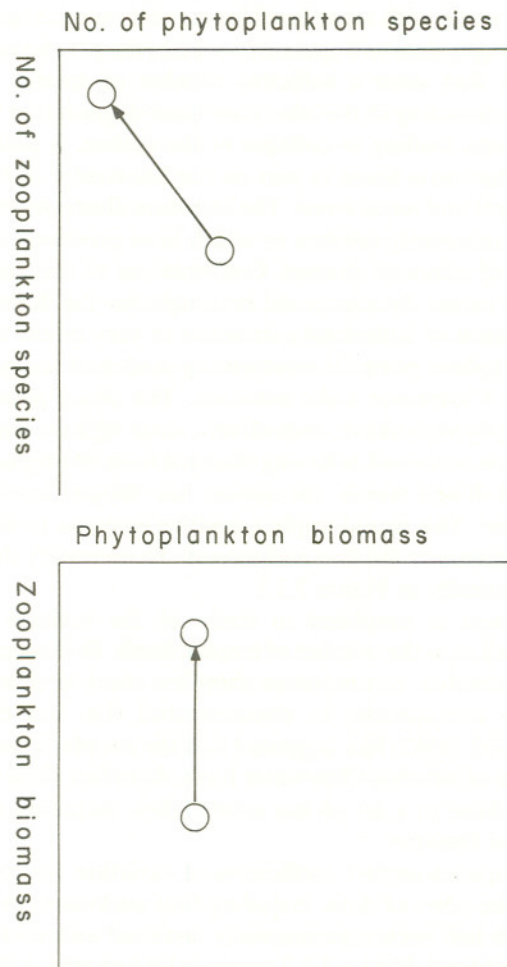


Figure 7.2.1 Schematics for the changes in numbers of species or in biomass for two interacting communities

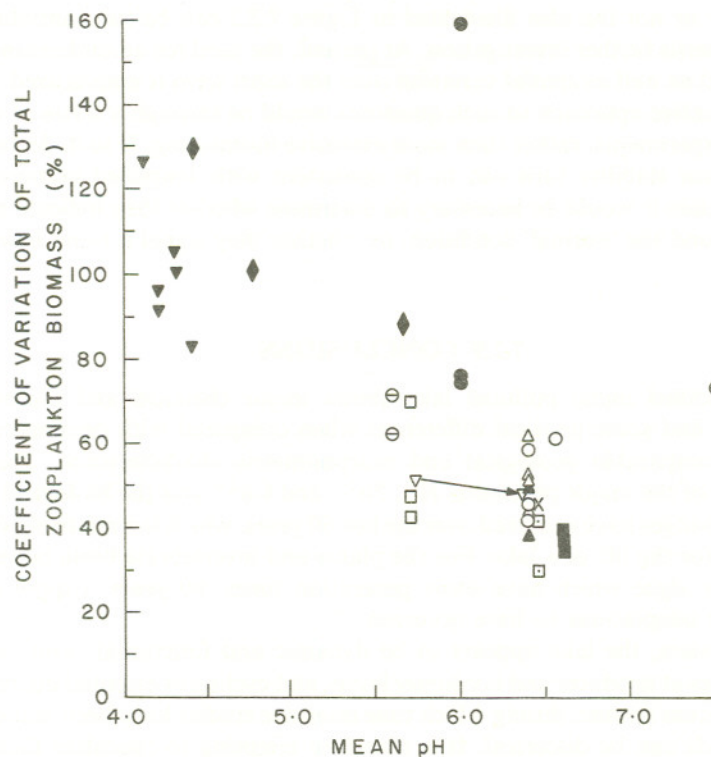
the extinction of fish populations alone would indicate that a return to the original state will not occur. Furthermore, massive changes in the soils and vegetation of the watershed, and residues of metals in sediments, make it unlikely that recovery will occur, at least over decades or even centuries. In view of this, it is irrelevant to the whole system to consider whether communities of lower organisms, plants, invertebrate animals and microorganisms, would 'recover', although, as the neutralization experiments suggested, the answer to this would be positive at least for the phytoplankton.

In answer to the second question, predictions on the future of the lake chemistry can be made with some confidence, since we can anticipate that the aluminium buffering system will operate to control the pH above 4.0. The present evidence suggests that since a sufficient number of species populations are functioning and interacting in the lake, continued degradation with further loss of functional groups, leading to collapse of the system, is unlikely in the near future. This assumption is based in part on our knowledge of the adaptation of organisms to low pH and metal stress. The literature illustrates many examples of species which are inherently tolerant or which have produced tolerant ecotypes for a wide range of chemical stresses. Confirmation of this relatively constant composition over recent decades could be sought for the diatoms at least, in a rather fine evaluation of community structure in very recent sediments.

Holling's (1973) phase portraits representing models of ecosystem change can be applied to the Clearwater Lake situation. The phase plane axes could be zooplankton and phytoplankton, respectively, since we have data for both. Very simply, the system is perceived as having changed from its original equilibrium to a new state. In Holling's words, the system has 'flipped from one domain of stability to another'. This would imply a positive response to the third question and a negative response to the first and second. To exemplify this, two cases are illustrated schematically in Figure 7.2.1.

This 'new' system is simplified in terms of the richness or diversity of populations, as well as in the number of trophic levels. In this state, it is pertinent to ask whether individual communities show less short-term temporal stability than comparable communities in circumneutral (i.e. less disturbed) lakes. Recently, Marmorek (1982) has suggested that the zooplankton communities in two acid lakes, one of which is Clearwater Lake, do in fact show more short-term fluctuation than those in a set of less acidic lakes, including some situated in South and Central Ontario.

Marmorek computed annual coefficients of variation for total zooplankton biomass for 33 lake-years of data, including four enclosures in a circumneutral lake, two of which had been experimentally fertilized and acidified, and two of which had been fertilized. Figure 7.2.2 supports his hypothesis that the more acid lakes show greater within-season fluctuations, in that the coefficients of variation are much higher for lakes with lower pH. The data set is limited, but the hypothesis (Marmorek, personal communication) has a sound theoretical basis:



Symbol	Lake	# Samples	Source	Symbol	Lake	# Samples	Source
▼	Clearwater	9-24	(1)	▲	Red Chalk	20-24	(1)
□	Chubb	23-24	(1)	◆	Cheat	13-14	(2,3)
▽	Nelson	8-9	(1)	⊕	Maggiore	9	(4)
●	Dickie	22-27	(1)	⊖	AF Cylinders	11	} this study
○	Harp	21-25	(1)	□	F Cylinders	11	
△	Jerry	16-24	(1)	X	Eunice	11	

Figure 7.2.2 Coefficients of variation for May to November, of total zooplankton biomass for 33 lake-years of data (after Marmorek, 1982)

'The coefficients of variation of total zooplankton biomass could vary inversely with pH because of 'holes' in the temporal organization of the communities, with no acid-tolerant species available with the appropriate life history and temperature response physiology to fill them.'

Interestingly, a preliminary analysis of the phytoplankton biomass data for a limited data set including Clearwater does not follow the pattern of pH-related coefficient of variation which was shown for the zooplankton.

Whether or not the idea illustrated in Figure 7.2.2 can be substantiated, it certainly merits further investigation. At present, the need for an understanding of temporal as well as spatial variation over the short term is emphasized. The most productive approach to such questions would be through a series of well-designed experiments, rather than more extensive monitoring. If such decreases in short-term stability turn out to be consistent with simplified community structure, then it would be necessary to determine whether they indicate mere 'noise' around the 'normal' condition, or whether they signal a trend towards extinction.

7.2.9 CONCLUSIONS

1. The acidified metal polluted lake shows major chemical and biological differences and some physical differences when compared with circumneutral lakes of comparable geological and morphometric characteristics. The introduction of the major pollutants H^+ , Ni^{2+} and Cu^{2+} and the biological and chemical changes have occurred over the last 60 years, which is a relatively short time span for the life of a lake. For the plants and invertebrate biota however, particularly algae which have short generation times, 60 years is quite long enough for adaptations to have occurred.
2. As a system, the lake appears to be dynamic and functional, with viable communities at producer and consumer levels, and cycling of essential nutrients. Over the recent decade, during which time intensive studies have been made, no major trends can be discerned. It is therefore tempting to speculate that the system as it is now is relatively stable. One significant factor in this stability is the existence of buffering systems which would maintain the pH in the mid-4 range and thus prevent major shifts in pH.
3. In terms of the specific pollutants which are related to the changes observed, pH seems to be the major controlling variable for the phytoplankton community and probably also for the benthos and microorganisms. For the zooplankton, pH is important but the metals are clearly additional controlling factors. The situation for fish is more difficult to assess; however, while the low pH *per se* in Clearwater is sufficient to explain the absence of fish, the metals were toxic even when the pH was raised by liming in Middle Lake, so that both types of stress are likely to be important. Experimental work, e.g. the type described for fish in enclosures, would be necessary to delimit precisely the role of H^+ , Cu^{2+} and Ni^{2+} for all of the communities.
4. Recovery to the original condition, in the absence of intervention, cannot be anticipated. One obvious reason for this is the massive damage and major changes sustained by the watershed of the lake. This observation emphasizes the importance of considering a lake ecosystem as the basin plus its surrounding watersheds, and also emphasizes the potential shortcomings of experimental manipulation such as lake or enclosure (limno-corrall) experiments to simulate

pollution 'in the field'. Such experiments have considerable value, but caution is needed in the interpretation, especially for long-term responses.

5. Recovery, in the sense of rehabilitation with the objective of restoring a viable fishery, also has to be addressed with guarded optimism. The experimental liming procedure may have been too 'coarse' but it achieved the desired objective in terms of chemical properties. However, the only community which 'recovered' rapidly in response to liming was the phytoplankton. Since other biota may require more time to adjust to the 'shock' of neutralization, it is important to monitor the long-term effects. This particular type of chemical treatment is, naturally, of great interest in view of the large number of lakes whose biota are endangered by acid stress.

6. While much remains to be explained, the approach which was used for the Sudbury Study, i.e. a comprehensive collection of data over an extended period of time, with reference or control sites included, is clearly essential for a study of a system under stress. Parallel experimental enclosures or microcosms should be considered as a valuable adjunct to the descriptive field studies.

7. In the future, it would be efficient and economical for studies on polluted systems if some general indicators or 'vital signs' could be identified in order to avoid the expensive and detailed collection of data. Alternatively, the dynamics of certain indicator species which have been shown to represent rate limiting steps in the over-all system may provide sufficient information to assess the condition of an ecosystem. The older concept of an indicator species, meaning one most sensitive to change, is in itself inadequate in the assessment process, unless there is compelling evidence that the species is also playing a key role in the system.

8. At the present time, on the basis of the Clearwater Lake study, it is not possible to recommend useful generalized vital signs or indicators. The concepts of instability and of high coefficients of variation in simplified systems merit further evaluation.

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