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Abstract

This study investigates the effects on waterfowl of ecological changes in aquatic ecosystems associated with acid precipitation. The distribution, density and habitat preferences of waterfowl were examined throughout much of north-eastern and central Ontario using aerial survey counts of indicated nesting pairs. The number of waterfowl breeding in acid-sensitive areas was estimated at 105 000 pairs. Breeding densities were relatively low (about one indicated pair per square kilometre), and no significant trends in population levels have been noted since 1980. Boreal species, including Common Goldeneye (*Bucephala clangula*), Hooded Merganser (*Lophodytes cucullatus*), Ring-necked Duck (*Aythya collaris*) and American Black Duck (*Anas rubripes*), commonly breed on small lakes, beaver flowages and wetlands most vulnerable to acidification, whereas Common Loon (*Gavia immer*) and Common Merganser (*Mergus merganser*) use large lakes and rivers. Dabbling ducks, including Wood Duck (*Aix sponsa*), Blue-winged Teal (*Anas discors*), Green-winged Teal (*Anas crecca*) and Mallard (*Anas platyrhynchos*), use well-buffered wetlands often associated with agricultural land.

In 1983, trophic relationships were studied on 123 small head-water lakes at different stages of acidification. Data on water quality, fish and macroinvertebrate community composition and waterfowl productivity were collected from an acid-sensitive area which receives moderate inputs of atmospheric pollutants (Ranger Lake) and a heavily acid-stressed area (Wanapitei) of north-eastern Ontario. The latter area receives a higher rate of acid deposition both from long-range transport processes and from smelting activities in nearby Sudbury, Ontario.

Lake environments were categorized by factor analysis into four major components of variation interpreted as axes of acidity (atmospheric deposition status), ionic strength (buffering status), eutrophication (wetland productivity status) and lake morphometry. Bedrock lithology explained much of the variability observed on the acidity and ionic strength axes and was a function of the balance between mineral acid inputs (SO_4^{2-}) and acid-neutralizing capacity (HCO_3^-, Ca^{2+} and Mg^{2+}). Weak organic acids did not contribute significantly to the acidity of the head-water lakes studied. Continued inputs of mineral acids into sensitive lakes in the Ranger Lake area will lead to further mean annual pH declines and pH depressions in shallow lakes during the spring melt-water period.

The availability of food for many waterfowl species has been influenced by lake acidification in the lakes we studied in north-eastern Ontario. An increased occurrence of fishless lakes in the acid-stressed area was correlated with lake acidity. Small, non-game fish species richness was primarily a function of lake acidity, although physical parameters (lake and drainage basin morphometry) also influenced fish species composition, especially within the cyprinid assemblage. Both piscivorous species studied (Common Loon and Common Merganser) used large lakes with large drainage basins. The number of broods produced in relation to the number of indicated nesting pairs observed was lower in the acid-stressed area than in the unaffected area. The Common Loon and Common Merganser are most at risk from acidification because of effects on fish.

In the absence of fish, in either acidic or non-acidic lakes, large, mobile macroinvertebrates, such as Notonectidae, Corixidae, *Graphoderus liberus* (Dytiscidae) and *Chaoborus americanus* (Chaoboridae), were abundant. Insectivorous waterfowl, e.g., Common Goldeneye and Hooded Merganser, preferred fishless lakes as a probable consequence of reduced competition with fish for common insect prey. Omnivores, e.g., Ring-necked Duck and American Black Duck, showed no preference for acidic or fishless lakes, but did prefer shallow, nutrient-rich wetlands. Further research to determine whether acidity ultimately limits the abundance of invertebrate prey of non-piscivorous waterfowl is under way.

Foreword

This report contains the results of research carried out under the auspices of the Long Range Transport of Air Pollutants programme, an interdepartmental research initiative of the federal government involving Agriculture Canada, Fisheries and Oceans Canada, Energy, Mines and Resources Canada, Health and Welfare Canada, and Environment Canada. Within Environment Canada, research into various aspects of long-range transport of air pollutants is being carried out by the Atmospheric Environment Service, Inland Waters/Lands, and the Canadian Wildlife Service (CWS).

The CWS research programme was started in 1980 to assess the impacts of acid deposition on wildlife and wildlife habitats in eastern Canada. The results of the first phase of the programme are contained in this and forthcoming volumes in the Occasional Papers series.

A major objective of the CWS research was to compare avian breeding and feeding ecology data collected from sensitive head-water habitats receiving different rates of acid loading. This first paper describes the work on waterfowl and their food-chains in Ontario, while a forthcoming one will include the results of surveys of freshwater bird communities in Quebec, as well as phyto-ecological studies of their associated habitats, in relation to acidification.

Other important areas of interest are the influence of long-range deposition and acidification on metal uptake by wildlife prey organisms and the toxicity of low-level metal exposure to aquatic birds. Long-range transport of airborne pollutants can affect the availability of heavy metals to biota both by direct transport and by the mobilization of metals from soils and sediments as acidity increases. Another forthcoming Occasional Paper will include preliminary results of research at the National Wildlife Research Centre on the fate of heavy metals in waterfowl food-chains, as well as laboratory studies of the effects of dietary heavy metals on the reproductive output of birds under controlled conditions.

Together these volumes will provide a summary of the first phase of the CWS LRTAP programme. The objective of this phase was to determine which species and habitats might be most at risk from acidification. Current studies are designed to establish a more definite cause-and-effect relationship between acidification and biological changes, chiefly in bird communities; to provide the basis for a biomonitoring programme which will track the changes expected to occur as emissions are reduced to the target loading (i.e., 50% of 1980 levels by 1994); and to evaluate the adequacy of that target loading for protecting aquatic biota.

Interdisciplinary studies of calibrated basins form an important aspect of the LRTAP programme. CWS has played a major role in one of these, the Kejimkujik Calibrated Catchment programme, studying nutrient release in and limnological characteristics of acidified waters in Kejimkujik in Atlantic Canada. Results of these and other related CWS studies on acidification are included in the *Final report of Impact Assessment Work Group 1 of the U.S.–Canada Memorandum of Intent* (1983); the two-volume proceedings of the International Symposium on Acidic Precipitation held at Muskoka, Ontario, in 1985, edited by H. Martin and published as Vol. 30 of *Water, Air and Soil Pollution* (1986); and the proceedings of an International Workshop on Birds as Bio-indicators held in Kingston, Ontario, in 1986 and published in *The uses of birds,* edited by A.W. Diamond and F.L. Filion, a Technical Publication of the International Council for Bird Preservation (Cambridge, U.K., 1987).

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Introduction

Acid precipitation has become a serious environmental threat to aquatic and terrestrial ecosystems in eastern North America, Scandinavia and other parts of Europe (Memorandum of Intent 1983). The deposition of atmospheric pollutants, such as sulphur and nitrous oxides, and their interaction with hydrological and soil processes in watersheds may result in significant alterations to surface water (lake/stream) chemistry, including decreased pH, increased trace metal concentrations and modified base cation, organic anion and nutrient relationships, as depicted in Figure 1. These chemical changes have been linked to biological changes (Dillon *et al.* 1984; Kelso *et al.* 1986*a*), including mortality of biota (Nyholm 1981; Magnuson *et al.* 1984; Campbell and Stokes 1985) and deterioration in habitat quality, often resulting in reproductive impairment, reduced growth and condition of adults and young, or behavioural changes (Nyholm and Myhrberg 1977; Ormerod *et al.* 1985). Such changes may alter the composition and abundance of several levels of the food-chain and affect predator–prey interactions (Eriksson *et al.* 1980; Haines 1981) (Fig. 1).

Despite evidence linking acid precipitation with damage to aquatic biota, including fish (Beamish and Harvey 1982; Beamish 1974; Schofield 1976; Magnuson *et al.* 1984), aquatic invertebrates (Weiderholm and Eriksson 1977; Campbell and Stokes 1985; Stephenson and Mackie 1986) and amphibians (Pough and Wilson 1977; Clark and LaZerte 1985; Freda and Dunson 1985), little is known about the effects of acidification on organisms that are not wholly aquatic (Clark and Fischer 1981). Because of their dependence on the aquatic environment for nest sites, brood protection and food, waterfowl may be seriously affected by acid precipitation (Haines and Hunter 1982; Eriksson 1984) (Fig. 2). The loss and degradation of aquatic habitat may have serious implications for the future of the waterfowl resource in eastern North America. Formerly secure waterfowl habitats in the forests of eastern Canada are now threatened by hydroelectric power and recreational developments, certain forestry practices, and industrial effluent pollution and atmospheric contamination (North American Waterfowl Management Plan 1986). Concern for the potential threat to the waterfowl resource prompted the Ontario Region of the Canadian Wildlife Service to undertake studies to determine effects of acid precipitation on waterfowl populations breeding in north-eastern Ontario. These studies have been designed to provide comparisons of waterfowl breeding success and food-chain relationships between areas receiving different rates of acid deposition, but with similar waterfowl breeding distributions and densities.

Figure 1
Diagram illustrating the probable pathway of effects of, and biological response to, the acidification of surface waters

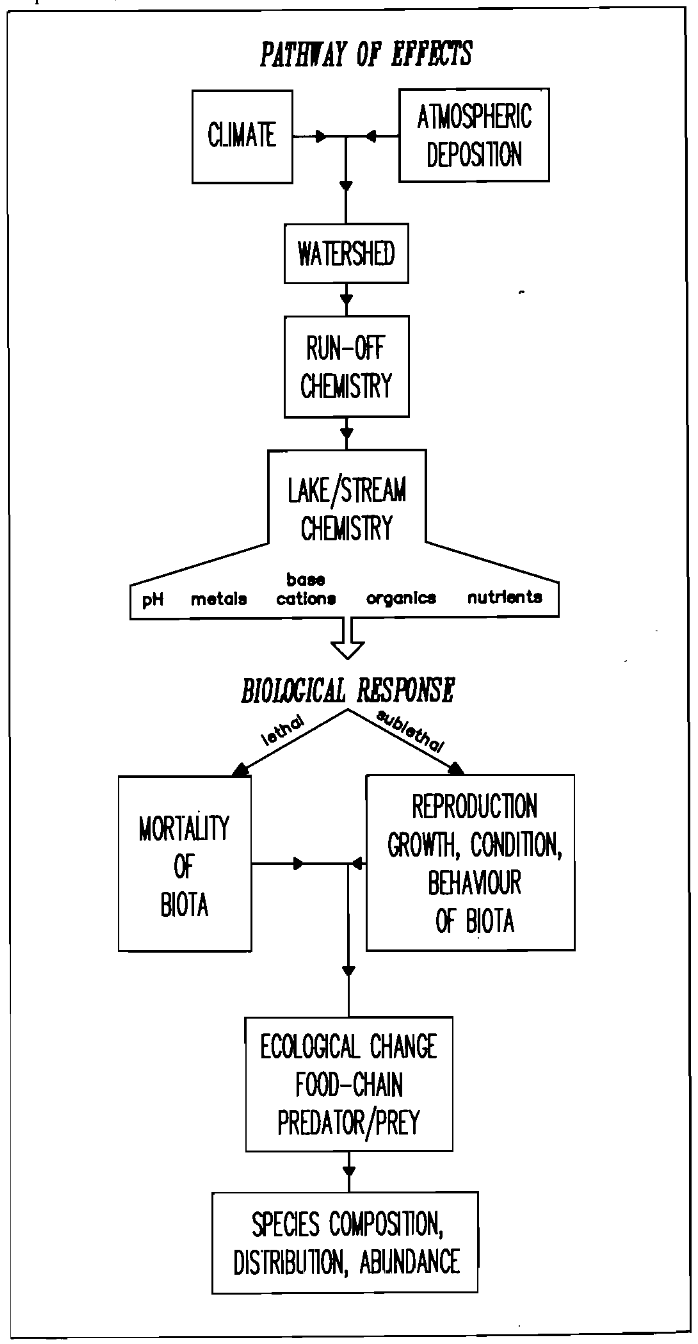
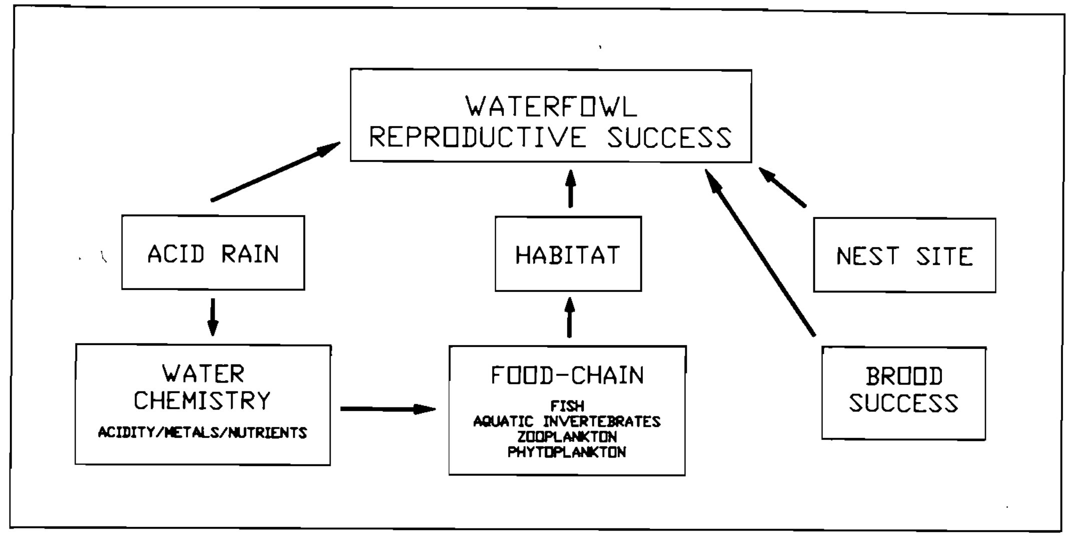


Figure 2
Diagram illustrating the complexity of habitat selection parameters which influence waterfowl reproductive success



Waterfowl, including the Common Loon (*Gavia immer*) in this study, were identified as particularly susceptible to acidification of the aquatic environment for the following reasons:

(1) They largely rely on aquatic and riparian habitat for both food and cover for nesting and brood-rearing.

(2) Females of certain species must acquire most of their resources for egg production near the breeding grounds and do not carry extensive fat reserves from wintering or staging grounds.

(3) Young, being praecocial, must forage in the immediate vicinity of the nest site and are not brought food by the adults.

(4) Their relatively large size necessitates a greater concentration of food resources and thus less latitude before declines in food become detrimental.

Increased acidity, combined with elevated trace metal concentrations, may lead to changes in predator–prey relations within the aquatic food-chain which, coupled with direct toxicological effects, could limit the availability, abundance and quality of waterfowl foods (Eriksson *et al.* 1980, 1986; Haines and Hunter 1982). Increased acidity has been linked to reduced food resources for certain waterfowl (DesGranges and Darveau 1985; McAuley 1986; DesGranges and Rodrigue 1986), other waterbirds in Sweden (Nilsson and Nilsson 1978) and Dippers (*Cinclus cinclus*) in Wales (Ormerod *et al.* 1985, 1986).

Acid deposition in Ontario

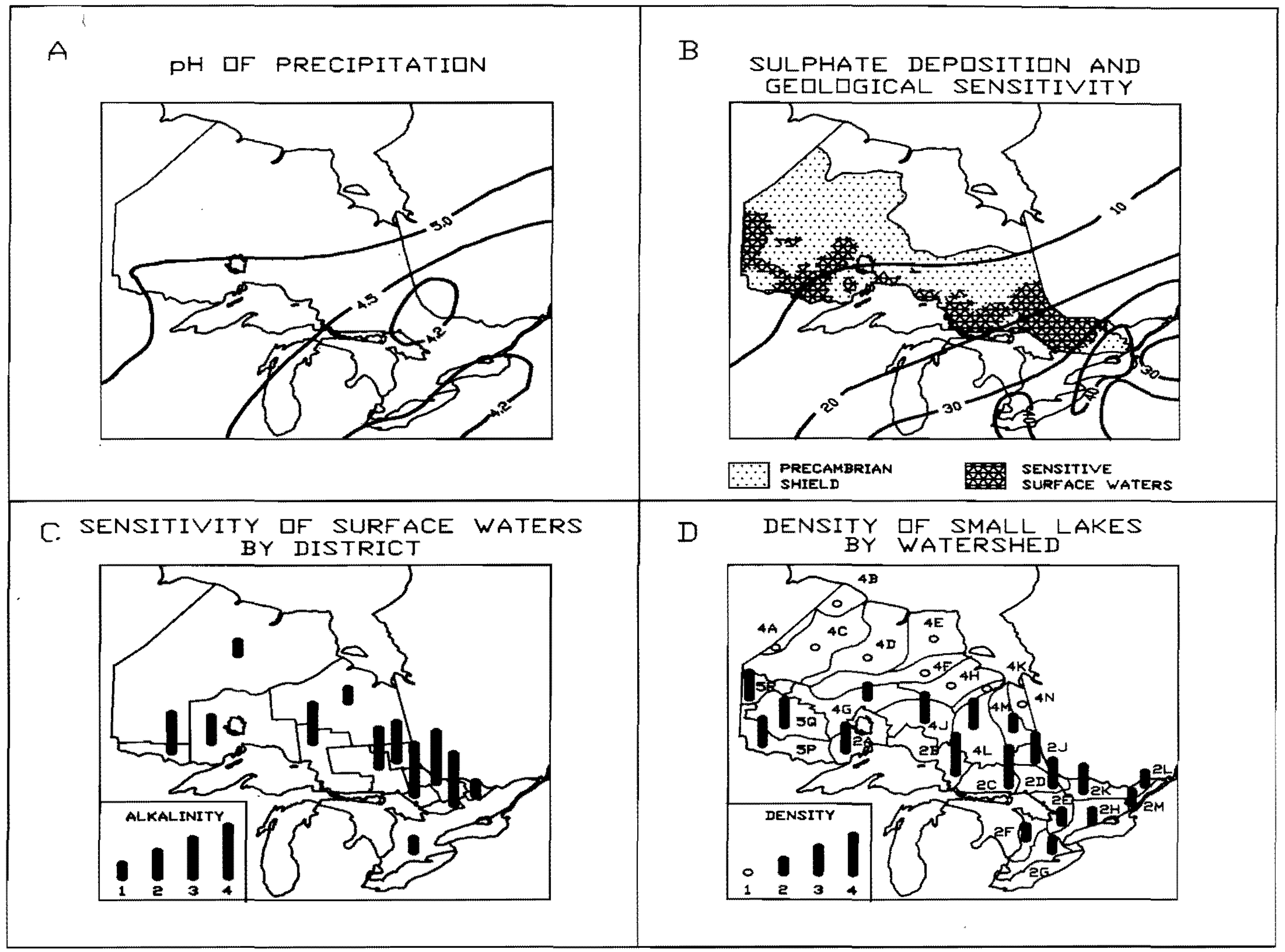
More than 700 000 lakes in eastern Canada receive acid deposition above background levels; more than two-thirds of this lake area is in Ontario (Kelso *et al.* 1986*a*). Much of north-eastern, central and southern Ontario receives substantial acid precipitation, with the average annual pH below 4.5 (Memorandum of Intent 1983) (Fig. 3*a*). Both the greater Sudbury area and the lower Great Lakes–St. Lawrence River valley corridor receive highly acidic precipitation, with the average annual pH in precipitation below 4.2. Sulphate is the dominant anion in precipitation and contributes most to the long-term acidification of surface waters. Annual wet sulphate deposition levels exceed 20 $kg \cdot ha^{-1} \cdot yr^{-1}$ for much of north-eastern and southern Ontario, with areas of south-western Ontario receiving in excess of 40 $kg \cdot ha^{-1} \cdot yr^{-1}$ (Memorandum of Intent 1983) (Fig. 3*b*).

The extent to which an area receiving acid precipitation will ultimately suffer surface water acidification depends on several factors. These include elevation (Verry 1981), the ratio of drainage basin area to lake area (Dillon *et al.* 1978), hydrologic characteristics of the terrain (Hendry *et al.* 1980; Eilers *et al.* 1983) and vegetation type and land-use management practices. The most important factor, however, is the geochemistry of the watershed (Kramer 1976; Kaplan *et al.* 1981; Brousseau *et al.* 1985). Although southern Ontario (south of 45°N Lat.) receives high levels of acid deposition, the lakes are well buffered against the adverse effects of acid precipitation by limestone bedrock that contains an almost infinite acid-neutralizing capacity. Much of north-eastern and central Ontario, however, is underlain by sensitive Precambrian bedrock covered by a thin layer of glacial moraine deposits (Fig. 3*b*). Streams, rivers and lakes in regions such as Muskoka, Haliburton, Parry Sound and Algonquin Park, as well as parts of Manitoulin, Sudbury, Timiskaming and Algoma Districts, receive atmospheric loadings several times greater than those in more northerly and westerly areas of the province; moreover, they offer little to neutralize acidic inputs and are therefore more likely to incur decreases in alkalinity and pH as a consequence.

Minns (1981) estimated that more than 300 000 ha, or approximately 11 400 lakes, fall into moderate- to high-risk categories. Of the 5341 lakes surveyed recently in Ontario (Ministry of Environment 1985), nearly 60% contained low acid-neutralizing capacity (defined as having alkalinities $< 200\ \mu eq \cdot L^{-1}$), and may be regarded as moderately to highly sensitive to acid precipitation (Fig. 3*c*). The most severely affected lakes are located in areas associated with the mining and smelting industries near Sudbury.

Among the most sensitive aquatic resources are those in the head-water portion of watersheds and small, enclosed water bodies (Haines 1981). Head-water systems may be expected to acidify more quickly than larger lakes, because their ionic chemistry is governed to a greater extent by precipitation chemistry and the hydrogeochemical composition of the watershed (Eilers *et al.* 1983). On the Precambrian Shield, small head-water lakes occur frequently and comprise a sizeable portion of the available aquatic habitat for waterfowl. Small lakes (<9.9 ha) are particularly abundant in geologically sensitive watersheds of north-eastern and central Ontario (Cox 1978) (Fig. 3*d*).

Figure 3
Maps of Ontario illustrating (*a*) precipitation amount-weighted mean annual pH in 1980 (from Memorandum of Intent 1983); (*b*) mean annual wet sulphate deposition ($kg \cdot ha^{-1} \cdot yr^{-1}$) (from Memorandum of Intent 1983) showing outline of Precambrian Shield and geologically sensitive areas which contain surface waters at risk from acidification (from National Research Council of Canada 1981); (*c*) percentage of lakes and streams with moderate to low alkalinity ($< 200\ \mu eq \cdot L^{-1}$) by county or district (from Ministry of Environment 1985) classed as follows: 1 = 0–25%, 2 = 26–50%, 3 = 51–75%, 4 = >75%; and (*d*) density (no. per 100 km^2) of small lakes (<9.9 ha) by secondary watershed units (from Cox 1978) classed as follows: 1 = 0, 2 = 1–20, 3 = 21–40, 4 = >40



Study framework

Head-water lakes represent significant nesting and brood-rearing habitat for many waterfowl species. The breeding ranges of several species, including Common Goldeneye (*Bucephala clangula*), Hooded Merganser (*Lophodytes cucullatus*), Ring-necked Duck (*Aythya collaris*), American Black Duck (*Anas rubripes*), Mallard (*Anas platyrhynchos*), Common Merganser (*Mergus merganser*) and Common Loon, overlap geologically sensitive regions of the province. Little is known about the distribution and habitat preferences of waterfowl breeding in northern Ontario, although nesting densities are generally considered to be low. Only one reconnaissance survey (Dennis 1974*a*) has examined the breeding distribution of waterfowl in north-eastern Ontario, but few of these plots fell close to areas in this study. We undertook systematic breeding pair surveys throughout the entire region, as well as more intensive surveys in the immediate vicinity of each study area. The objectives of the survey component of this study were as follows:

(1) To quantify waterfowl breeding density throughout north-eastern Ontario so as to (*a*) calculate the number of waterfowl breeding in acid-sensitive areas that are potentially susceptible to habitat degradation, and (*b*) examine trends in species distribution across the region;

(2) To examine temporal variations in species abundance;

(3) To determine basic patterns of habitat selection, particularly with respect to wetland size, and to compare these patterns between the two study areas.

To develop a framework in which to study the effects of acid precipitation on waterfowl, a pilot study was conducted from 1980 to 1982 in a moderately stressed, but largely unaffected, area of north-eastern Ontario (Ranger Lake) to determine:

(1) The chemical and physical parameters, as well as biotic interactions, that govern the quality of waterfowl breeding habitat;

(2) Predictions of how surface water acidification would alter breeding habitat suitability, and ultimately the reproductive success of waterfowl species.

As a consequence of these studies on waterfowl populations, water chemistry and food-chain interactions, we hypothesized that the adverse effects of lake acidification on waterfowl would be mediated through changes in the abundance and availability of prey species.

Local extinctions of fish populations have occurred in acid-sensitive areas of eastern North America (Beamish and Harvey 1972; Schofield 1976; Pfeiffer and Festa 1980; Watt *et al.* 1983) and Scandinavia (Wright and Snekvik 1978; Muniz and Leivestad 1980). Swedish workers (Eriksson *et al.* 1980) have postulated that many of the biological changes observed in acidifying lakes result from altered predator–prey relations as a consequence of the loss of fish populations. The disappearance of fish would create an environment suitable for the proliferation of more acid-tolerant invertebrates. Such a shift from a fish- to an invertebrate-dominated trophic system would have clear implications for waterfowl species that preferentially breed on small ponds, temporary pools, beaver flowages and head-water lakes. Studies conducted in 1983 were designed to test the following predictions in an area that has undergone considerable acid loading (Wanapitei), by drawing comparisons with an area currently undergoing acidification (Ranger Lake).

Hypothesis: The reproductive performance (brood production) of waterfowl in small head-water lakes undergoing acidification depends directly on food resources that can be altered and/or lost as a consequence of that acidification.

Linkages:

(1) The reduction in overall fish stocks at moderately high pH levels (>5.5) would adversely affect piscivores, e.g., Common Loon and Common Merganser.

(2) The loss of fish would lead to the proliferation of acid-tolerant invertebrates to the short-term benefit of insectivorous waterfowl, e.g., Common Goldeneye and Hooded Merganser.

(3) As acidity increased, invertebrates would also decline, adversely affecting insectivorous species as well.

Study areas

1. Description of the Ranger Lake study area

In 1980, a pilot study was initiated in the Ranger Lake area in the Algoma District, north-east of Sault Ste. Marie, Ontario (46°55′N Lat.; 83°35′W Long.) (Fig. 4). Ranger Lake was selected because it currently receives moderate amounts of acid deposition (Figs. 3*a*, 3*b*) which can be attributed to long-range transport processes. The area is typical of north-eastern Ontario, and contains terrestrial and aquatic ecosystems relatively free from human disturbance such as logging and cottage development. It is underlain by early Precambrian granitic bedrock with low to insignificant buffering capacity (Fig. 3*b*). Ranger Lake lies within the Great Lakes – St. Lawrence lowland forest region (Rowe 1972), and is characterized by mixed hardwood forests, although elements of the boreal forest extend into the northern portion of the study area.

Located 50 km to the north-west of Ranger Lake is the Turkey Lakes Watershed (TLW) calibrated basin (Fig. 4). The Turkey Lakes Watershed is one of five calibrated watersheds in eastern Canada where detailed benchmark monitoring studies of long-range pollutant deposition are in progress. Data on terrestrial, aquatic and atmospheric quality, as well as biological response, are being collected to understand how these ecosystems function and respond to acid rain. The Turkey Lakes Watershed area receives a moderate to high level of acid deposition. Estimates of wet sulphate inputs have ranged from 20 to 35 kg·ha^{-1}·yr^{-1} (Barrie *et al.* 1982; Thompson and Hutton 1985). Annual mean precipitation pH values for the period 1981–83 were 4.43, 4.31 and 4.25, respectively (Semkin *et al.* 1984). Similar deposition levels would be expected at Ranger Lake. Ranger Lake has the additional advantage of having a large number of head-water lakes conducive to waterfowl studies.

Figure 4
Map of north-eastern Ontario showing the location of the eight systematic survey blocks and the Ranger Lake special block, the Ranger Lake (VIXEN and GONG) and Wanapitei study areas, and the Turkey Lakes Watershed (TLW) calibrated basin

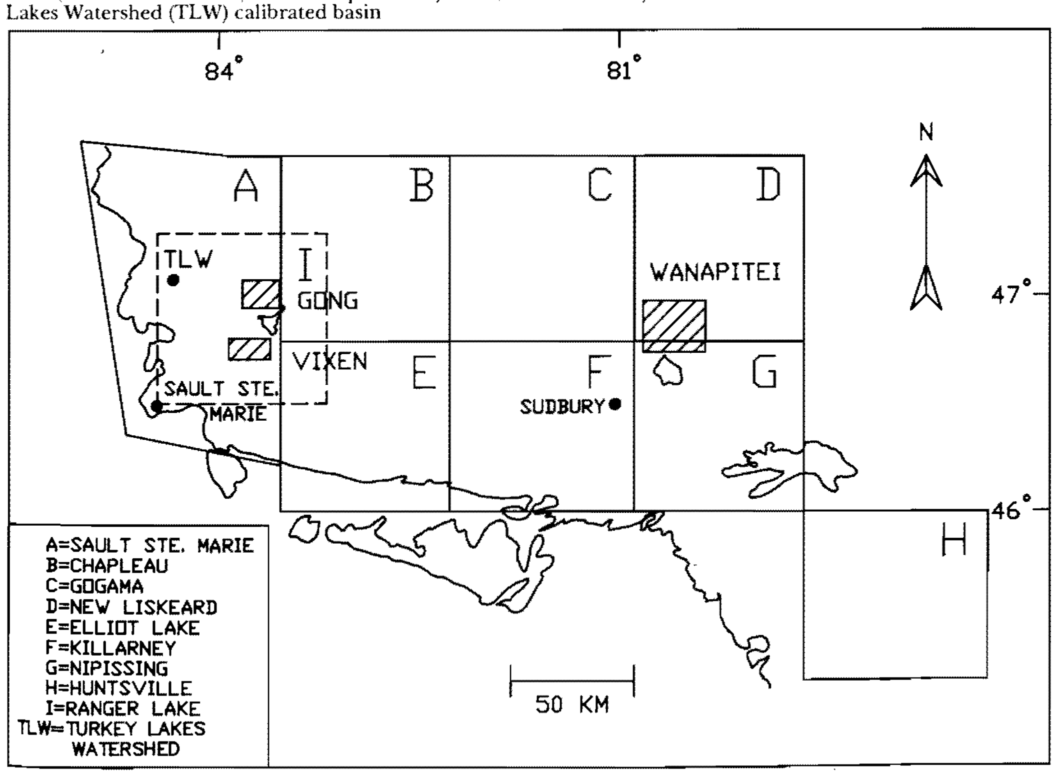
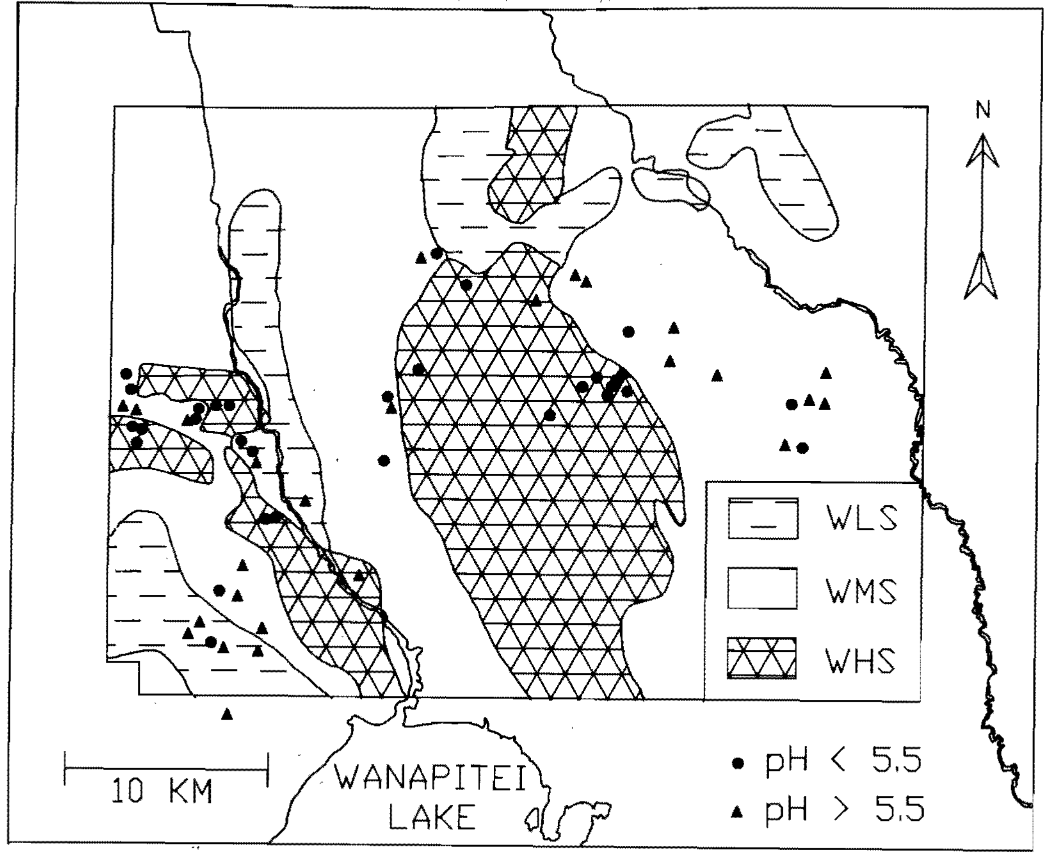


Figure 5
Map of the Wanapitei study area showing the outline of the 12 townships used in selective surveys and the location and lake pH class (acid <pH 5.5 > non-acid) of study lakes in relation to the bedrock sensitivity groups (high (WHS), moderate (WMS) and low (WLS) sensitivity)



In the Ranger Lake area, two major vegetational and geological zones were studied (Fig. 4). The VIXEN area, located south-west of Ranger Lake, contains a mixed hardwood forest dominated by yellow birch (*Betula alleghaniensis*) and sugar maple (*Acer saccharum*), with white pine (*Pinus strobus*) and white spruce (*Picea glauca*) dominant on mineral soil sites. In this area lacustrine outwash, deltaic and end moraine deposits, rich in calcium salts, override the general sensitivity of the underlying granitic bedrock (App. 1). These surface materials are often different from the underlying bedrock because they have been transported long distances from their source (Cowell *et al.* 1980; Brousseau *et al.* 1985). As a result, the buffering potential of streams and lakes associated with the Goulais River system is enhanced (Boissonneau 1968). The GONG area, north-west of Ranger Lake, lacks any additional buffering potential and is highly sensitive to acidification (App. 1). The area contains boreal forests dominated by black spruce (*Picea mariana*), white spruce, balsam fir (*Abies balsamea*) and white birch (*Betula papyrifera*). A portion of this area was substantially altered by a forest fire in 1969.

Both the GONG and VIXEN study areas cover parts of three different drainage systems. VIXEN lakes drain via the Goulais River (watershed code 2BF-3) towards Lake Superior or via the Garden River (2CA-2) towards Lake George. GONG lakes are drained either by the Goulais River (2BF-3) or by the Aubinadong River (2CB-2), which drains south into the North Channel. A total of 3186 lakes is contained within the three watersheds studied (App. 2) (Cox 1978). Of these, 81% are less than 9.9 ha in size. The average density of small lakes in the watersheds studied is 54.9 per 100 km^2, which is comparable with that found throughout the Eastern Lake Superior Tributaries watersheds (2B) (Fig. 3*d*).

2. Description of the Wanapitei study area

2.1. Atmospheric deposition and lake acidification

In 1983, concurrent studies were conducted in two areas of north-eastern Ontario receiving different acid loading levels: Ranger Lake and Wanapitei. As previously described, Ranger Lake contains sensitive aquatic ecosystems that currently receive moderate acid deposition stress, but whose surface waters remain largely non-acidic (pH > 5.5). The Wanapitei study area, located north-east of Sudbury (46°45′N Lat.; 80°45′W Long.) (Fig. 4), contains lakes whose surface waters range from heavily stressed and acidic (pH < 5.5) to unstressed (pH > 5.5) (Fig. 5).

Although Ranger Lake and Wanapitei are geographically quite close (within 225 km), the latter has received much higher mean annual inputs of sulphate deposition compared with Algoma (Fig. 3*b*). In their review of sulphate yields in lakes in relation to sulphate deposition in eastern Canada, Thompson and Hutton (1985) showed that recent estimates of wet sulphate deposition in Algoma (Turkey Lakes Watershed, 1980) were substantially lower than at Sudbury (1976), 20 vs. 37 kg·ha^{-1}·yr^{-1}, although recent estimates (1981–84) show a much higher current rate of input to Algoma (25–35 kg·ha^{-1}·yr^{-1}) than previously thought. The greater Sudbury area has had a history of fumigation problems, dating back to the early 1900s. Mining, smelting and lumbering activities have caused extensive alterations of the environment surrounding Sudbury, including contamination of soils and lake waters with trace metals (Hutchinson and Whitby 1977) and the reduction in pH and elevation of SO_4^{2-} levels in many lakes and ponds (Conroy *et al.* 1974).

Wanapitei lies within the large zone of high sulphate deposition which extends to the north-east and south-west of the sulphide ore smelting centres in Sudbury. The emissions of sulphur dioxide (SO_2) ranged from 4240 to 7034 t·day^{-1} for the period 1960–69. Since the mid-1970s, a number of factors, including the construction of the 381-m "superstack," smelting process changes, pollution abatement measures and extended strike and shut-down periods in the Sudbury metal recovery industry, have resulted in substantially reduced airborne emissions (Keller and Pitblado 1986). During the period 1978–83, estimated annual SO_2 emissions from the Sudbury smelters ranged from 1065 to 2562 t·day^{-1}, whereas during the 1970–77 period annual SO_2 emissions ranged from 3663 to 6383 t·day^{-1}.

In response to the threat of permanent damage to Sudbury area ecosystems, the Ontario Ministry of the Environment mounted the Sudbury Environmental Study (SES) in 1973. During the period 1974–76, a survey of 209 lakes within a 250-km radius of Sudbury was conducted to document the influence of atmospheric deposition on lake waters on a regional basis and to provide a data base to predict future trends. These surveys revealed that many lakes to the north-east and south-west of Sudbury were acidic (pH < 5.5) and had suffered reduced salmonid fisheries (Conroy *et al.* 1974; Pitblado *et al.* 1980), similar to problems encountered elsewhere in North America and Scandinavia. Dry deposition of SO_2 and sulphur fumigation, as well as deposition rates of SO_4^{2-} and the trace metals copper (Cu), nickel (Ni), zinc (Zn) and iron (Fe) to Sudbury area lakes, were found to be highly dependent on the distance from the smelter stack. Precipitation chemistry monitored during the June 1978 to May 1979 period showed that collecting stations north of Sudbury received, on average, lower annual bulk deposition levels of SO_4^{2-} (34 kg·ha^{-1}·yr^{-1}) than stations near Sudbury (46 kg·ha^{-1}·yr^{-1}) (Jeffries 1984). Measurements for these lakes were only slightly higher than similar measurements taken in Muskoka–Haliburton and other Precambrian Shield areas (Scheider *et al.* 1979), although substantially lower than historical levels. In the immediate vicinity of Sudbury (within 40 km), local emissions contribute up to 70% of the total Ni and Cu deposited (Chan *et al.* 1984), although declines in trace metal emissions have accompanied recent reductions in SO_2 emissions (Keller and Pitblado 1986). The deposition rates of aluminum (Al), manganese (Mn), nutrients and major ions (except SO_4^{2-}) were similar or only slightly higher at stations north of Sudbury than rates in south-central Ontario (Muskoka–Haliburton) (Jeffries 1984). Therefore, to minimize the effect of local deposition of metals (Cu, Ni, Zn and Fe) and dry deposition of SO_2 in the present study, study lakes were selected between 38 and 70 km from the smelter at Copper Cliff (Fig. 4), yet well within the zone of low pH lakes extending north-east of Sudbury.

A survey conducted between 1981 and 1983 to resample the 209 lakes originally sampled in 1974–76 revealed that substantial improvements in water quality had taken place in the intervening period (Keller and Pitblado 1986). Observed water quality changes included increases in pH and decreases in SO_4^{2-}, Ni and Cu concentrations. In one extremely acidic lake close to the smelter, the pH increased from 4.05 in 1972 to 5.8 in 1984, while concentrations of SO_4^{2-}, Cu, Ni, Co, Mn and Zn in the lake water decreased by 60–90% during the same period (Hutchinson and Havas 1986). The degree of observed changes showed a general relationship to distance from the Sudbury smelter, indicating that the nearly 50% reduction in contaminant deposition from Sudbury sources was responsible for the observed recent improvement in water quality (Keller and Pitblado 1986).

2.2. Geomorphology and hydrology

Situated at the convergence of the Superior, Southern and Grenville formations of the Precambrian Shield and two major fault systems, the greater Sudbury basin has a geomorphology that is exceedingly complex. Plutonic, metavolcanic and metasedimentary types of silicate bedrock are found in the area. Within the lake basins studied, bedrock types range from felsic end members (quartzite) through rocks of intermediate composition (volcanics, gneiss, migmatite) to carbonate-rich siliceous sedimentary rocks (App. 1). Although igneous and metamorphic rocks predominate north-east of Sudbury, substantial variability in the acid-neutralizing capacities of surface waters exists (Conroy *et al.* 1978). The relative insolubility of felsic and intermediate groups contrasts with the occurrence of carbonate-rich siliceous sedimentary rocks, often coupled with an ample cover of post-glacial sediment. Such bedrock and surficial geology contributes to the enhanced buffering potential of surface waters within certain lake basins. Bedrock sensitivity ratings for individual lake basins were determined at Ranger Lake (Map 1549A) and Wanapitei (Map 1550A) (App. 1) using Geological Survey of Canada maps, which depict the sensitivity of bedrock and derived soils to acidic precipitation in south-central and south-eastern Canada (Shilts 1981). Additional information was obtained from Ontario Geological Survey maps of bedrock (Map 2419) and surficial (Map 5465) geology. As in Brousseau *et al.* (1985), each lake was classified according to its sensitivity and then compared with the bedrock and surficial geology type that formed its watershed. Lakes with watersheds containing several geological types were classified according to the least sensitive rock type.

At Wanapitei, three distinct geomorphological regions were defined according to these bedrock sensitivity ratings (Fig. 5), as follows:
(1) LOW SENSITIVITY (WLS): carbonate-rich siliceous sedimentary rocks.
(2) MODERATE SENSITIVITY (WMS): mafic to intermediate metavolcanic rocks.
(3) HIGH SENSITIVITY (WHS): felsic intrusive and metamorphic rocks.
Over much of the area characterized by moderately or highly sensitive bedrock types, surficial deposits are composed of a thin overburden of silty to sandy till (App. 1). Thicker end moraine deposits of sand and gravel occur in the low-sensitivity region, further enhancing the residual acid-neutralizing capacity of the area (Fig. 5).

Five watersheds of the French River system (2D) are contained within the area studied (App. 2). The Upper Wanapitei River (2DA-2), East Wanapitei River (2DA-3) and Parkin Creek (2DA-4) drain into Lake Wanapitei directly, whereas the Chiniguichi (2DC-2) and Sturgeon (2DC-1) Rivers drain into Georgian Bay via Lake Nipissing and the French River. A total of 2152 lakes is contained within these watersheds, of which nearly 80% are less than 9.9 ha in area (App. 2). The average density of small lakes in the Upper

Wanapitei system (2DA) (55.1 per 100 km^2) is roughly the same as Ranger Lake (54.9 per 100 km^2). Fewer small lakes occur in the Sturgeon River system (2DC), yet the overall density (37.3 per 100 km^2) remains higher than most secondary watersheds in the Precambrian Shield (Fig. 3*d*).

To support the open roasting methods of early smelting, the forest surrounding Sudbury was logged heavily in the early 1900s. Regeneration has resulted in a poorly developed, second-growth forest characterized by jackpine (*Pinus banksiana*), white spruce, balsam fir and white birch.

Unlike many regions of central Ontario, human disturbance from cottage development is notably lacking in both the Ranger Lake and Wanapitei study areas. Because of their geographic proximity, factors such as migration habits, mortality levels, nesting habitat selection and breeding chronology would likely not differ between the two study areas.

Methods

1. Aerial survey design and schedule

1.1. Systematic surveys

To examine patterns of waterfowl population density and distribution, a systematic survey was instituted throughout north-eastern Ontario (as shown in Fig. 4). The region was divided into the standard universal transverse mercator (UTM) blocks of 100 km per side; the irregularly shaped Sault Ste. Marie block results from combining a number of partial blocks along a UTM grid convergence. Within each block, a survey plot (2 × 2 km) was laid out in the south-west corner and then every 20 km to the north and east for a total of 25 plots per block. In the convergence block, plots were selected that had the same easting and northing designations as those of a normal block. Only those plots falling on mainland were covered. A special block (see Fig. 4) employing the same grid of plots was designated in the Ranger Lake area. Schedules for coverage of systematic survey blocks are summarized in Table 1.

Table 1
Schedules for coverage of systematic survey blocks

Block	Date	Coverage
Ranger Lake (Special)	10–16 May 1980	Plots 1–19
	11–15 May 1981	Plots 20–25
	14–18 May 1985	Total
Killarney	7–9 May 1981	Total
	10–14 May 1985	Total
Gogama	8–10 May 1981	Total
	13 May 1985	Total
Huntsville	5–6 May 1981	Total
New Liskeard	11 May 1985	Total
Nipissing	9 May 1985	Total
Elliot Lake	13–15 May 1985	Total
Chapleau	13–18 May 1985	Total
Sault Ste. Marie	13–18 May 1985	Total

1.2. Selective surveys

At Ranger Lake, 94 plots (2 × 2 km) were laid out in 1980 to cover head-water lakes with known water chemistry. In 1981, 1982 and 1985, 43 of these plots (even numbers) were surveyed to examine annual variation in duck populations. An additional 44 plots were located around wetlands for coverage in 1981. These surveys yielded data on 649 wetlands (rivers excluded); survey results for 1981 were used in this data set along with those from 1980 for wetlands not covered in 1981.

At Wanapitei, surveys were restricted to a zone encompassing 12 townships north of Lake Wanapitei (Fig. 5). A random selection of all wetlands that fit within a plot of 2 × 2 km was made in each township so that approximately equal numbers of wetlands were chosen from each. A total of 414 wetlands was surveyed in 1983, some of which were selected for more intensive ground studies of potential mechanisms of food limitations to developing waterfowl.

1.3. Habitat description

Basic habitat characteristics of all wetlands surveyed in the study areas were determined through interpretation of aerial photographs (scale 1:15 840). Parameters include area of open water, total shoreline length, and length of shoreline with well-developed littoral zone (e.g., marsh, belt of ericaceous plants). Shoreline development indices (shoreline length divided by the square root of the area of open water) were calculated from these measurements.

1.4. Field procedure

All surveys employed a common field procedure detailed in Ross (1985) and summarized as follows. The surveys were undertaken from a Bell 206 B helicopter equipped with a range extender on the fuel tank and bubble windows on the back doors. These windows allowed the observers to extend their heads approximately 25 cm outside the limits of the original body of the aircraft so that visibility was greatly enhanced.

An observer sitting in the front passenger seat of the helicopter acted as navigator and data recorder and alerted the other observers to upcoming birds. These observers sat at the back of the aircraft by each window and notified the navigator of waterfowl sightings through an intercom system. The species, sex, number and exact location of all birds seen were recorded directly on acetate-covered aerial photographs on which the location of the wetlands and the boundaries of the plots were drawn. For this study, a wetland was defined as any body of open water visible on an aerial photograph (scale 1:15 840). Individual bodies of standing water (lakes, ponds, beaver flowages and sloughs) were usually self-evident; however, where they were irregularly shaped or in a series along a drainage, each was delineated by the presence of a clearly visible outlet stream. Streams and rivers were usually treated as separate wetlands unless they were very small, in which case they were considered part of the nearest lake or pond with which they were associated. Small interconnecting streams were divided among the associated standing water bodies. Occasionally wetlands had undergone considerable change since the photography and, in some cases, new ones had been created by beaver activity. A note was made during the survey and the new shoreline sketched on the aerial photos. The aircraft passed over all wetland habitat at altitudes as low as 20 m above the ground and speeds ranging from a hover to 100 km·h^{-1}. Multiple passes were made over some wetlands where the presence of birds was suspected or where the sex and species of the birds could not be ascertained on the first pass. Because of the small, discrete nature of boreal forest wetlands on the Precambrian Shield, total coverage of the habitat was accomplished by following the shorelines.

1.5. Timing

Surveys were undertaken during a limited period or "survey window" when most of the local breeders had started nesting. This period begins when the migrants have passed through the area and before the desertion by the males during incubation (as in Dzubin 1969 and Dennis 1974*a*). The extent of this period in northern Ontario is not known, as no detailed phenology studies are available for that area. These surveys have used a conservatively short period of two weeks, starting one week after the smallest wetlands (< 10 ha) were free of ice (early to mid-May). As only one survey flight was made, it was not always possible to sample each species within its appropriate "window." Population density estimates for late-nesting species, particularly the Ring-necked Duck, should be considered approximate. They nest almost one month after most other species, although their actual migration appeared to be complete prior to the survey flights.

1.6. Data analysis

Results are expressed in numbers of "indicated pairs" per species, which is based on the number of lone males, pairs and males in flocks of five males and fewer (as in Dzubin 1969). The sex of the Black Duck could not usually be determined in the field, and so "indicated pair" estimates were generated using the known sex ratios of the closely related Mallard as in Dennis (1974*a*). The following multiple regression equation was developed using all breeding pair survey data available for Ontario wetlands where Mallards were found:

$$Y = 0.0700 + 0.632\,X_1 + 1.1166\,X_2 + 0.7398\,X_3 \quad [1]$$
$$R^2 = 0.9577,\ p < 0.00001$$

where Y = number of indicated pairs of Mallards
X_1 = number of lone Mallards
X_2 = number of flocks of two Mallards
X_3 = number of Mallards in flocks between 3 and 10

Indicated pair estimates for Black Ducks can then be generated by substituting Black Duck sighting information for that of Mallards in the equation. Wetlands which had no Black Ducks ($X_1 = X_2 = X_3 = 0$) were assumed to have no indicated pairs ($Y = 0$), i.e., the equation was not employed under these circumstances.

The Common Loon is also effectively monomorphic. Because these birds are very strongly territorial, indicated pairs were determined from the presence of either a single bird or a pair in proximity. For the Canada Goose (*Branta canadensis*), indicated pairs were counted for each single bird, pair or flock of three, whereas larger flocks were ignored.

Breeding densities are expressed as a number of indicated pairs per 100 km^2 and are accompanied by a standard deviation estimate. This has been generated using a formula developed by G.E.J. Smith for systematic samples (Ross 1987).

2. Limnology, waterfowl utilization and trophic studies

2.1. Pilot studies (1980–82)

The methods used in the pilot study undertaken in the Ranger Lake area between 1980 and 1982 are described below with reference to the comparative studies conducted in 1983. Where these methods differ, further elaboration is contained in Appendix 3.

Ground studies conducted in the two areas in 1983 included water sampling and waterfowl surveys, as well as minnow and aquatic invertebrate sampling. Small, shallow head-water ponds and lakes, draining relatively small areas, were studied (Table 2). Lakes ranged in size from 0.6 to 27.2 ha (median = 5.0 ha) and were generally shallow (1.0–30.5 m, median = 4.4 m).

Table 2
Lake morphometry parameters used in study lake selection, expressed as medians with range in parentheses

Study area	N	Lake area, ha	Maximum depth, m	Drainage area, ha	Hydrologic types I–II	Hydrologic types ≥III
Ranger Lake	69	5.50 (0.7–26.3)	4.59 (1.0–30.5)	189.8 (16.3–2078.0)	41%	59%
Wanapitei	55	4.39 (0.6–27.2)	4.39 (1.0–14.0)	70.4 (2.7–1855.0)	66%	34%

All statistical analyses were performed using the Statistical Analysis System (SAS) computer procedures (Statistical Analysis System Institute Inc. 1982).

2.2. Water quality

Water sampling was conducted between 5 and 9 June at Wanapitei and between 12 and 17 June at Ranger Lake. Prior to sampling, temperature profiles were measured at the deepest location in the lake. Following Ministry of Environment (1979) guidelines, a non-weighted, composite water sample was collected, using flexible plastic Tygon tubing lowered through the epilimnion and metalimnion in stratified lakes, or to a depth of 1 m above the bottom in shallow ponds or lakes. Where maximum depths did not exceed 1 m, grab samples were collected. Following preservation, samples were cooled to 4°C and transported to the Great Lakes Forestry Research Centre in Sault Ste. Marie, Ontario, for chemical analyses. All samples underwent coarse filtering using a Wheaton No. 4 filter to remove suspended particulate matter. Samples prepared for nutrient and carbon analyses underwent further filtering using a 0.45-μm millipore (Sartorius) filter.

Chemical determinations, procedures and equipment, given in the Department of the Environment (1979) guidelines, were followed (App. 4), except for total inflection point alkalinity (HCO_3^-), which was measured by an electrometric titration method equivalent to the Gran technique. Within 24 h of collection, conductivity (μmho·cm^{-1} at 25°C) and pH were determined potentiometrically. Conductivity measurements were standardized to 20°C and corrected for hydrogen ion concentration (KCORR) using methods presented by Sjors (1950). Analyses were completed for other major ions, including calcium (Ca^{2+}), magnesium (Mg^{2+}), potassium (K^+), sodium (Na^+), sulphate (SO_4^{2-}), chloride (Cl^-) and silica (SiO_2); nutrients including total Kjeldhal nitrogen (TKN), ammonia (NH_3), nitrite–nitrate (NO_2 + NO_3), total phosphorus (TP) and soluble reactive phosphorus (PO_4); total carbon (TC), total organic carbon (TOC) and total inorganic carbon (TIC); and trace metals, including Al, Mn, Zn, Cu, Ni, lead (Pb), cadmium (Cd) and Fe. Sulphate determinations were made using the methyl thymol blue (MTB) method. True and apparent water colour determinations (expressed in APHA Platinum–Cobalt Colour Units) were made on samples collected in both the littoral zone (< 1 m) and mid-lake from 24 June to 5 July 1984 at Ranger Lake and 25 July to 8 August 1984 at Wanapitei.

2.3. Habitat description

As in section 1.3., data were also collected on physiographic characteristics of lakes and their watersheds. Estimates of lake surface area (LA), shoreline length (SL) and riparian area (RIP), which was calculated as the area within the forested edge minus the open water area, were derived from interpretation of aerial photographs (scale 1:15 840). Wetland ratios (WETRAT) were calculated as the ratio of riparian area to lake surface area. Shoreline development indices (SDI) were also calculated. Maximum lake depths (ZMAX) were determined from bathymetric data collected for each lake. For Wanapitei study lakes, the distance to the Copper Cliff smelter was measured. Lake elevation (ELEV) and the total upstream drainage area (DRAIN) of each lake were measured from topographic maps (scale 1:50 000).

Topographic maps and aerial photographs were used to define hydrologic types of lakes according to drainage order position (see Table 2) and the relative magnitude of surface water exchange through inlets and outlets (modified from Eilers *et al.* 1983) as follows:
(1) Type I: no permanent inlets or outlets (ground-water inflow may be very small or moderate).
(2) Type II: intermittent outlet only (ground-water inflow may be moderately strong).
(3) Type III: permanent inlet and outlet (reflecting relatively strong surface water through-flow).
(4) Type IV: more than one permanent inlet and outlet (reflecting downstream drainage order hierarchy and strong surface water through-flow).

2.4. Minnow sampling

Estimates of small, non-game fish species occurrence and abundance were obtained using baited wire minnow traps. Fish sampling was conducted between 30 June and 3 August 1983. The cylindrical traps were constructed of 6-mm wire mesh and measured 1.0 m by 0.30 m with 40-mm openings in each conical end. A trap, baited with dog biscuits, was located in shallow water (<2.5 m) for 24 h at each of five sites equidistant around the shoreline of each lake. Each trap was suspended from floats and anchored to prevent drifting, except in shallow water where the traps often rested on the bottom. Captured minnows were preserved in 10% formalin prior to sorting, identification and counting. Other organisms found in the traps, including tadpoles, newts, crayfish and macroinvertebrates, were noted and released.

2.5. Aquatic invertebrate sampling

Aquatic macroinvertebrate communities were examined on a subset of 18 lakes in each area. Lakes were chosen across the available range of pH, included those with and without fish, and ranged in size from 1.5 to 7.5 ha.

A standardized sweep net procedure was used to compile lists of invertebrate taxa for comparative purposes. Ten samples, consisting of 10 consecutive sweeps, were taken at equidistant points parallel to the shoreline of each lake in water less than 1 m deep. Each sweep was taken over the bow of a forward-moving canoe and described an arc from the water surface to as near the substrate as possible and back to the surface. The net used had a bag 43 cm deep with 9 meshes per centimetre, on a D-frame with 625 cm^2 area, and a 122-cm handle. Invertebrates were sampled between 17 and 24 July at Wanapitei and between 26 July and 1 August in the Ranger Lake area.

Insects were identified to as low a taxonomic level as possible, given the life stage that made up the major part of each taxon. The only exception was Corixidae, for which identification has been made only on adults because immatures, which made up a large portion of many samples, could not be adequately identified. Insect groups not included in the analyses were those of terrestrial origins and the aerial adults of aquatic forms, especially Diptera, which were often taken from the water surface. Each lake was scored for the presence or absence of major insect taxa, which were defined as those recorded in sweep net samples from more than one lake in an area.

2.6. Waterfowl surveys

Distributions and densities of breeding adult waterfowl were determined from helicopter surveys conducted during the nest-initiation period in May 1980 and 1981 for Ranger Lake and May 1983 for Wanapitei (as described in section 1.2.). Regular ground surveys were conducted on all water bodies in 1983 to assess adult and brood activity. Each water body was surveyed twice during the brood-rearing period, once between 19 June and 5 July, and a second time between 26 July and 7 August. Surveys were conducted concurrently in both study areas using two teams of two trained observers each. Surveys were conducted between 0600 and 1200 h. Water bodies were not surveyed under extreme wind or rain conditions. Survey teams undertook complete canoe and/or ground checks of all aquatic and riparian habitat. The location, number and age of young were noted on scale maps of the study lakes, along with observations on foraging and other behaviour. Ducklings were aged according to the Gollop and Marshall (1954) plumage development scheme.

Results and discussion

1. Waterfowl breeding distribution in north-eastern Ontario

1.1. General patterns

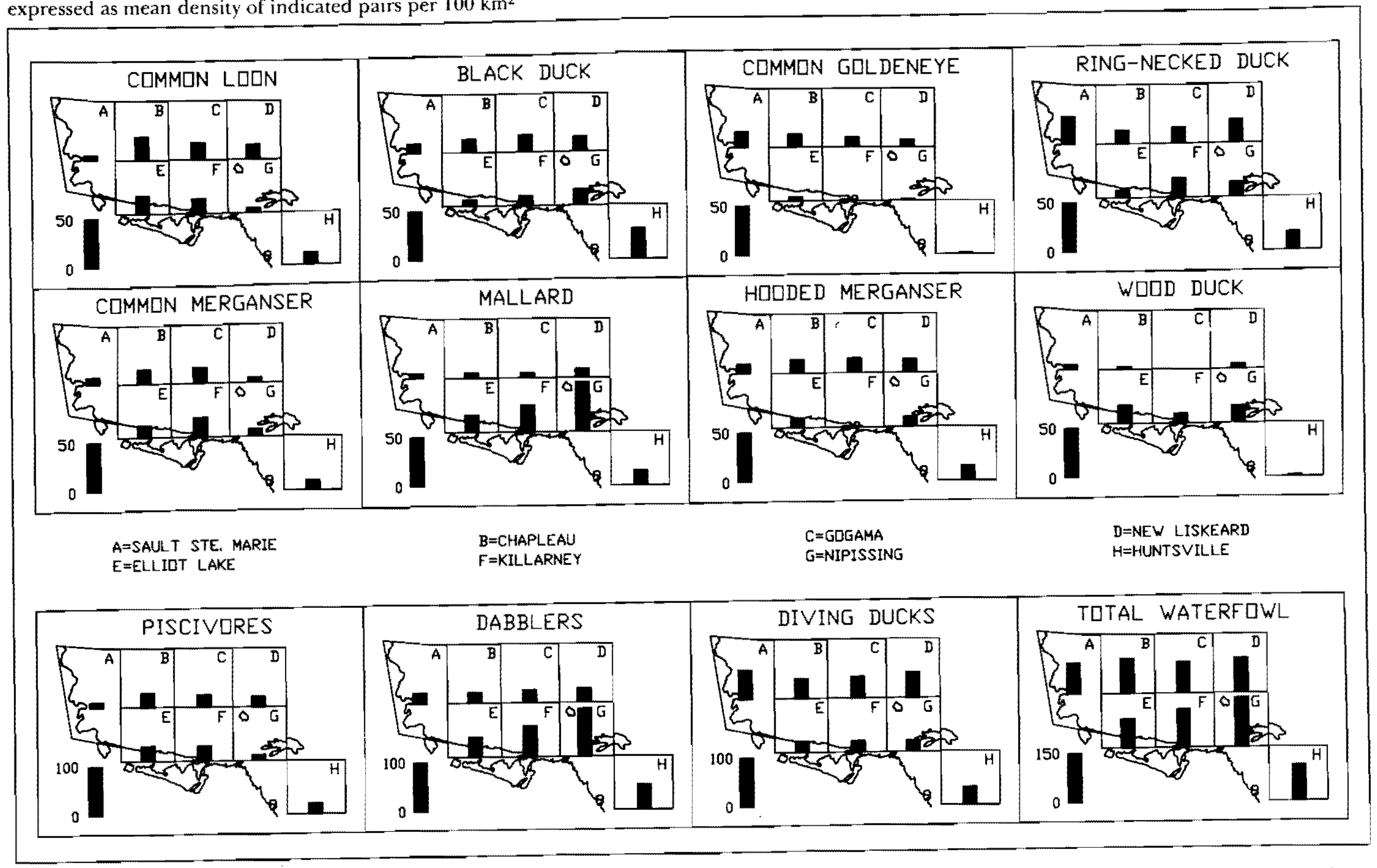
The north-eastern Ontario survey blocks (Fig. 4) extend over a zone of intergradation between Great Lakes – St. Lawrence and boreal forest regions (Rowe 1972). Throughout the region, the amount of wetland habitat was surprisingly consistent, ranging from 4.9 to 6.0 wetlands per plot (Table 3). The northern tier of blocks, including the Sault Ste. Marie convergence block, which best represents the VIXEN and GONG study areas, are physiognomically quite similar. They are dominated by hilly boreal terrain, with patchy coniferous forest and lakes of various sizes evenly distributed throughout. Overall waterfowl breeding densities are quite similar in these blocks (about one pair per square kilometre) (Fig. 6). The southern tier of blocks extends over more deciduous habitat, except in the Nipissing block, where much of the area is low-lying and flat with more extensive swampy wetlands and agricultural influence. The superior habitat in the Nipissing block is reflected by the highest overall breeding densities recorded anywhere, whereas other southern blocks hold numbers similar to those of more northerly blocks (Fig. 6).

Patterns in the geographic distribution of certain waterfowl species and taxonomic groups are evident in Figure 6, and are described in the annotated list of species

Table 3
Breeding densities of waterfowl in the eight blocks surveyed in north-eastern Ontario, expressed as mean density and standard deviation of indicated pairs per 100 km^2

Block:	Sault Ste. Marie		Chapleau		Gogama		New Liskeard		Elliot Lake		Killarney		Nipissing		Huntsville		Average	
Number of plots:	28		25		25		25		19		23		25		25			
Wetlands/plot:	5.20		5.40		4.92		4.52		6.00		5.30		5.72		5.76			
Species	Dens.	± SD	Dens.	± SD	Dens.	± SD	Dens.	± SD	Dens.	± SD	Dens.	± SD	Dens.	± SD	Dens.	± SD	Dens.	± SD
Common Loon	3.57	2.09	22.00	4.20	17.00	3.85	13.00	3.43	18.42	5.71	15.22	5.22	5.00	2.67	11.00	4.34	12.82	1.39
Canada Goose	0.89	0.72															0.13	0.10
Mallard	6.25	3.00	6.00	2.84	6.00	3.05	9.00	6.46	17.11	7.04	26.09	9.29	52.00	13.66	14.00	4.94	16.80	2.51
American Black Duck	9.64	3.62	12.00	3.15	18.60	4.77	16.80	4.34	5.53	2.25	12.61	4.99	16.40	4.35	30.80	6.71	15.54	2.56
Northern Pintail													1.00	0.96			0.13	0.12
Green-winged Teal			2.00	1.93	2.00	1.24	1.00	0.96			6.52	4.74	10.00	3.56	4.00	2.36	3.20	0.84
Blue-winged Teal	0.89	0.72					2.00	1.47			7.61	4.28	11.00	7.04	3.00	2.22	3.08	1.09
Wood Duck	6.25	3.28	2.00	1.93			4.00	2.22	19.74	5.81	10.87	3.96	18.00	4.62	1.00	0.96	7.31	1.13
Ring-necked Duck	28.57	6.82	13.00	4.41	16.00	4.34	22.00	9.84	7.89	2.91	20.65	6.59	15.00	6.12	19.00	7.73	18.20	2.34
Lesser Scaup	3.57	2.87															0.51	0.41
Common Goldeneye	16.96	5.43	13.00	6.04	10.00	4.24	9.00	4.94	3.95	1.84			1.00	1.11			7.05	1.40
Bufflehead			2.00	1.11	1.00	0.96	3.00	2.89			1.09	1.06			2.00	1.47	1.15	0.47
Hooded Merganser	9.82	3.79	13.00	3.73	15.00	4.75	14.00	2.78	10.53	4.24	1.09	1.22	11.00	2.72	16.00	5.64	11.41	1.36
Common Merganser	8.04	3.04	14.00	3.60	7.00	3.20	6.00	2.36	13.16	4.69	15.22	5.33	8.00	3.47	10.00	3.56	10.00	1.29
Red-breasted Merganser	0.89	0.88											1.00	0.96			0.26	0.18
Total	95.36	16.77	99.00	15.40	92.60	15.70	99.80	21.80	97.63	17.79	116.96	15.02	149.40	26.75	110.80	19.09	107.59	6.74
% wetlands occupied	34.3%		39.3%		43.9%		45.1%		42.1%		41.8%		42.7%		37.5%		40.8%	

Figure 6
Breeding densities of eight common waterfowl species and combined taxonomic groups in the eight blocks surveyed in north-eastern Ontario, expressed as mean density of indicated pairs per 100 km^2



presented in Appendix 5. Dabblers, in particular Mallard and Wood Duck (*Aix sponsa*), were abundant in the southern tier of blocks, especially the Nipissing block. Diving ducks, in particular Common Goldeneye and Hooded Merganser, were more common in the boreal forests of the northern tier of blocks, including Sault Ste. Marie. In the Huntsville block, patterns of waterfowl distribution were similar to those in the northern tier of blocks, with the exception that Goldeneyes are notably absent. Piscivore numbers increased from east to west within the survey region, although numbers were lowest near the Lake Superior shore.

Using a modified reciprocal averaging ordination technique (TWINSPAN; Hill 1979*a*), an analysis of species and plot associations was performed on indicated nesting pair data from 191 systematic survey plots throughout the eight survey blocks. The resulting dendrogram (Fig. 7) illustrates the major associations among the 11 waterfowl species that average greater than one indicated nesting pair per 100 km^2 in the region. Piscivorous waterbirds (i.e., Common Loon and Common Merganser) often co-occurred in plots, although the Common Merganser was often found on rivers and streams. Typical boreal waterfowl species, including Common Goldeneye, Hooded Merganser, Ring-necked Duck and Black Duck, often occurred together on plots. Of the two remaining groups, the most significant is the Wood Duck, Blue-winged Teal (*Anas discors*) and Mallard group (Fig. 7), which is often associated with agricultural areas in the survey region.

An estimate of the number of breeding waterfowl potentially at risk from the effects of acid precipitation was obtained using data from surveys conducted in the region and acid sensitivity information (Fig. 7). As most, but not all,

Figure 7
Dendrogram produced by TWINSPAN illustrating major associations of waterfowl species from systematic surveys conducted in the eight blocks in north-eastern Ontario, and depicting the relative numbers of indicated pairs breeding in geologically sensitive areas receiving wet sulphate deposition in excess of 10 kg·ha^{-1}·yr^{-1}

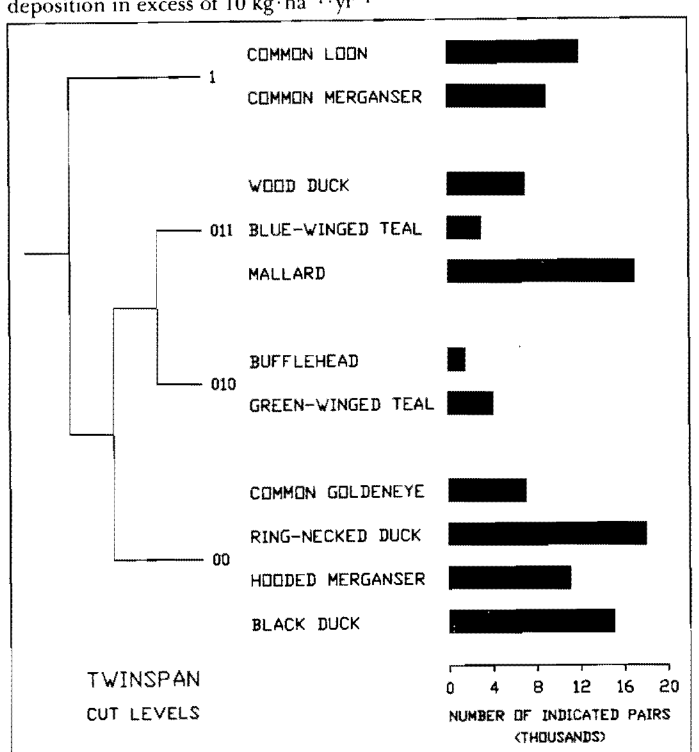


Table 4
Inter-year comparison of waterfowl survey results from three blocks in north-eastern Ontario, expressed as mean density of indicated pairs per 100 km^2

	Ranger Lake (Special)		Gogama		Killarney	
Species	1980–81	1985	1981	1985	1981	1985
Common Loon	9.00	10.00	21.00	17.00	13.04	15.22
Mallard	2.00	4.00	6.00	6.00	30.44	26.09
American Black Duck	9.60	6.00	19.40	18.60	12.17	12.61
Green-winged Teal			2.00	2.00	2.17	6.52
Blue-winged Teal					10.87	7.61
Wood Duck		2.00			4.35	10.87
Ring-necked Duck	13.00	24.00	27.00	16.00	21.74	20.65
Common Goldeneye	12.00	15.00	14.00	10.00	1.09	
Bufflehead	1.00	1.00	6.00	1.00		1.09
Hooded Merganser	10.00	15.00	14.00	15.00	9.78	1.09
Common Merganser	9.00	10.00	9.00	7.00	8.70	15.22
Total	65.60	87.00	118.40	92.60	114.35	116.96

Table 5
Comparison among four years of waterfowl surveys of 43 plots in the Ranger Lake area, expressed as mean density of indicated pairs per 100 km^2

Species	1980	1981	1982	1985
Common Loon	29.0	46.0	42.0	39.0
Mallard	9.0	17.0	9.0	18.0
American Black Duck	23.8	30.6	27.4	40.2
Green-winged Teal	1.0	2.0	0.0	0.0
Blue-winged Teal	0.0	0.0	1.0	1.0
Northern Shoveler	0.0	3.0	0.0	0.0
Wood Duck	0.0	6.0	8.0	8.0
Ring-necked Duck	40.0	45.0	36.0	44.0
Common Goldeneye	52.0	66.0	57.0	56.0
Bufflehead	2.0	3.0	1.0	1.0
Hooded Merganser	23.0	19.0	28.0	22.0
Common Merganser	11.0	14.0	33.0	25.0
Total	190.8	251.6	242.4	254.2

geologically sensitive terrain of north-eastern and central Ontario was surveyed, estimates of breeding populations have been extrapolated from existing information (Table 3). Approximately 105 000 pairs of waterfowl nest in sensitive habitat, covering roughly 97 500 km^2 of north-eastern Ontario that is currently receiving more than 10 kg SO_4^{2-}·ha^{-1}·yr^{-1} (Figs. 3*a*, 3*b*). Substantial numbers of piscivores (22 200 pairs), diving ducks (36 800 pairs) and dabbling ducks (44 700 pairs) are potentially affected (Fig. 7). Estimated totals for pairs of the more common species are: Common Loon 12 500, Common Merganser 9700, Wood Duck 7100, Blue-winged Teal 3000, Mallard 16 400, Bufflehead (*Bucephala albeola*) 1100, Green-winged Teal (*Anas crecca*) 3100, Common Goldeneye 6900, Ring-necked Duck 17 700, Hooded Merganser 11 100 and American Black Duck 15 100.

1.2. Temporal variability in waterfowl abundance

Three blocks (Ranger Lake, Gogama and Killarney) were resurveyed after four years (1981 and 1985) (Table 4), and showed considerable comparability in the relative abundance of species between the two surveys. Although there was occasionally some divergence in values for a given species, no statistically significant differences could be discerned (Wilcoxon signed ranks test) (Siegel 1956). Surveys based on a single visit have an added component of variability, as the waterfowl move among a number of lakes in their home ranges and are not necessarily on their nesting lakes when surveyed; it is entirely possible that a bird that breeds on the plot is temporarily off it when surveyed, or vice versa. The fact that the mean annual waterfowl density of the three blocks was virtually identical in the two years (1981, 99.45; 1985, 98.85) strongly suggests that sampling error plays the major role in the differences at the species level. These results are largely confirmed by the multiple surveys of the Ranger Lake area (Table 5), in which no significant trend could be identified. The low total count of waterfowl for 1980 may have resulted in part from inexperience, as this was the first operational year for the survey technique.

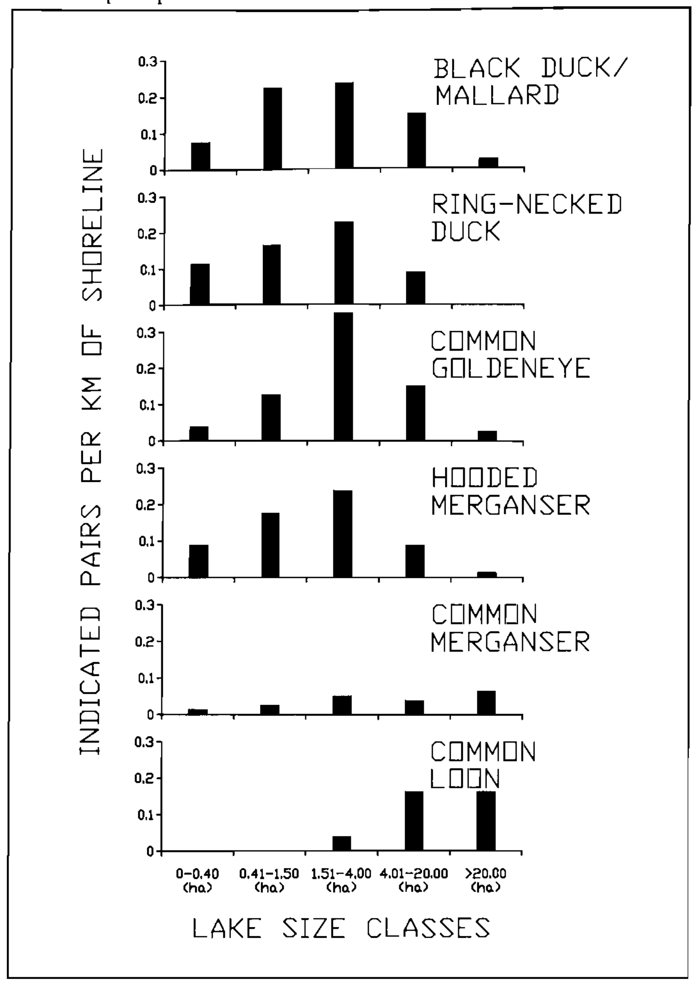
Taking all resurvey data together, individual waterfowl populations showed no consistent increase or decrease over several years. Only the Wood Duck showed a consistent increase for all comparisons. Although statistical significance cannot be attained, this trend is likely real, as this species has increased in numbers consistently in southern Ontario (Ross *et al.* 1984) and elsewhere in its range (Bellrose 1978). Other species had relatively stable populations, making it reasonable to compare abundance and distribution of these birds in different areas at different years within the study period.

1.3. Waterfowl distribution in the study areas

As the study was moved into a second, more acid-stressed area (Wanapitei) in 1983, it was necessary to determine if waterfowl were selecting similar habitats in the two areas, i.e., if the occupancy proportions of comparable lakes were alike. To do this, lakes were divided into various classes or cells according to overall area of water, shoreline development index and length of well-developed littoral zone (see App. 6 for cell descriptions). Occupancy of lakes (presence of an indicated pair) in similar cells could then be compared between the areas. These results from the 649 and 414 wetlands in the Ranger Lake and Wanapitei areas, respectively, are provided in Appendix 6 for six common species. Actual frequencies of occupied versus unoccupied lakes were compared through multiple χ^2 tests using the Mantel–Haenszel procedure for combining a set of 2 × 2 tables. These tests revealed no significant differences between the two study areas in the occupancy of comparable wetlands. Although relatively low breeding density makes statistical tests problematic, inspection of the values also indicates a considerable degree of similarity between the two areas. These results suggest that waterfowl are behaving similarly in response to the broad habitat types in the two study sites and so provide the continuity needed to make more generalized conclusions about brood habitat selection patterns and predator–prey relationships in both study areas.

Because no difference could be discerned in lake type selection, the data can be pooled for the two areas to examine lake size preference. Breeding density was calculated based on the number of indicated pairs of each

Figure 8
Breeding densities of six common waterfowl species in lakes of various size classes in the two study areas combined, expressed as the number of indicated pairs per kilometre of shoreline



species per kilometre of shoreline of lakes in the five size classes used in the previous analysis (Fig. 8). Clearly, insectivores (Common Goldeneye and Hooded Merganser) and generalists (Black Duck/Mallard and Ring-necked Duck) prefer small lakes, usually under 20 ha and particularly in the 1.5- to 4.0-ha range. Piscivores (Common Merganser and Common Loon) occupy larger water bodies, although Common Mergansers also used river and stream habitats. These results support the decision to restrict the study to smaller, primarily head-water, wetlands (see Table 2) to focus on as many individual waterfowl as possible.

2. Description of lake environments

2.1. Factor analysis

Basic statistics (mean ± SD, median) for 31 morphometric and chemical variables are summarized in Table 6, according to the five bedrock sensitivity groups defined in Appendix 1 (Ranger Lake — VIXEN and GONG; Wanapitei — high (WHS), moderate (WMS) and low (WLS) sensitivity). To describe the environment of the head-water lakes studied, a principal factor analysis was applied to 22 environmental variables. The variables were selected to reflect lake and basin morphometry (lake area, shoreline development indices, wetland ratio, cumulative drainage basin area and maximum detected depth), major ion chemistry (pH, KCORR, (Ca^{2+} + Mg^{2+}), K^+, Na^+, HCO_3^- and SO_4^{2-}), trace metal concentrations (Al, Mn, Cu, Zn and Fe) and nutrient levels (TOC, TKN, NH_3, NO_2 + NO_3 and PO_4). Those variables with highly skewed distributions or missing data were rejected, as indicated in Table 6. To normalize the distributions, the data were log transformed (except pH) and converted to unit scores so that each variable would be weighted equally for comparative purposes in the analysis. Only lakes with complete chemical,

Figure 9
Diagram illustrating the results of the principal factor analysis on the combined Ranger Lake and Wanapitei data set (N = 123 lakes). The percentage of individual factor variability explained by each variable is shown

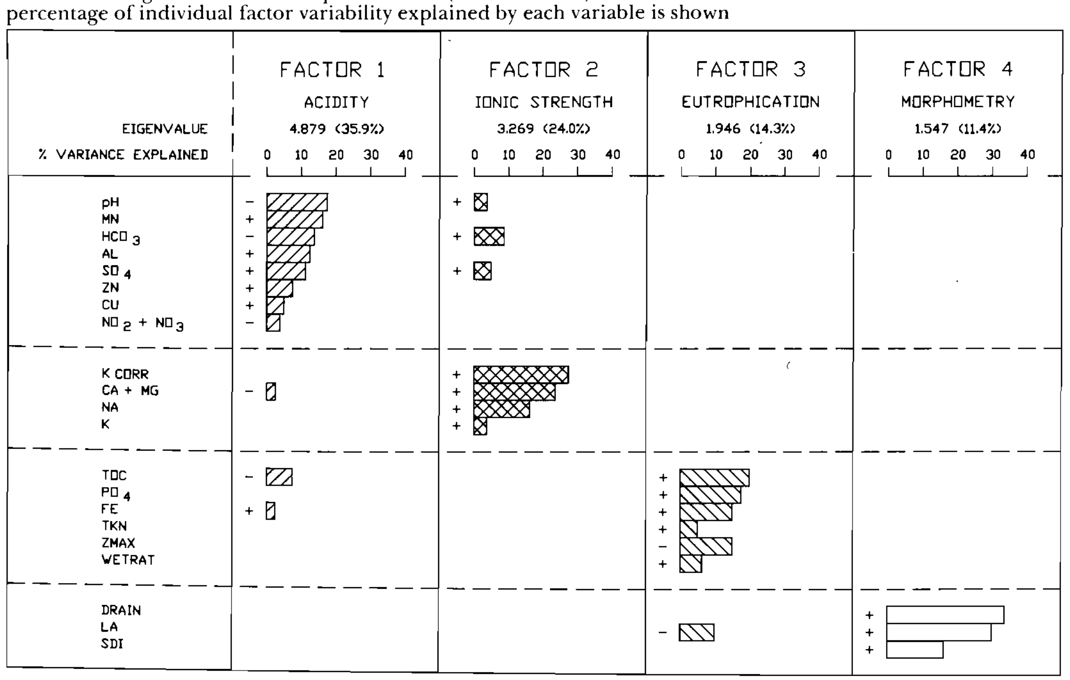


Table 6
Summary statistics (mean ± SD, median) for 31 chemical and physical parameters in lakes sampled in the Ranger Lake and Wanapitei study areas in 1983, according to bedrock sensitivity groups (Ranger Lake—VIXEN and GONG; Wanapitei—high (WHS), moderate (WMS) and low (WLS) sensitivity)

			Wanapitei									Ranger Lake					
			WHS (N = 20)			WMS (N = 24)			WLS (N = 11)			VIXEN (N = 35)			GONG (N = 34)		
Variable	Units		Mean	± SD	Median	Mean	± SD	Median	Mean	± SD	Median	Mean	± SD	Median	Mean	± SD	Median
Lake morphometry																	
1. Lake elevation	ft		1098	79	1050	1040	94	1000	1096	61	1100	1374	153	1450	1582	103	1600
2. Shoreline length	m		1108	566	1080	1947	1304	1708	1584	725	1786	1514	965	1188	2009	1243	1607
3. Lake area	ha	*	4.24	3.08	3.15	8.68	8.33	4.55	6.75	4.85	7.0	6.59	6.28	4.60	9.74	6.73	8.25
4. Riparian area	ha		1.97	1.63	1.50	4.37	6.60	2.35	1.91	2.04	1.90	2.71	6.25	1.10	2.79	5.91	1.30
5. Wetland ratio		*	1.62	0.63	1.48	1.69	0.74	1.53	1.51	0.67	1.28	1.55	0.78	1.33	1.34	0.47	1.17
6. Shoreline development		*	1.57	0.34	1.55	2.00	0.84	1.78	1.82	0.34	1.73	1.78	0.45	1.66	1.82	1.19	1.69
7. Basin area	ha	*	122.8	130.4	63.4	296.1	464.9	81.3	74.1	40.5	74.6	445.4	524.9	230.7	256.8	235.1	160.9
8. Maximum depth	m	*	4.36	2.69	3.40	5.31	3.58	4.50	6.38	4.24	5.20	6.24	5.80	4.60	6.64	6.31	4.50
Lake chemistry																	
9. pH		*	4.75	0.40	4.65	5.50	0.67	5.69	6.09	0.89	6.33	6.57	0.49	6.66	6.17	0.43	6.28
10. Corrected conductivity	μmho·cm⁻¹	*	27.3	4.9	27.8	31.5	6.6	31.1	39.2	14.0	34.2	33.7	9.2	34.8	23.0	3.3	23.3
11. Water colour	Hazen units		14	13	10	25	18	20	25	17	20	44	32	38	25	16	20
12. Alkalinity (TIP)	μeq·L⁻¹	*	3.4	12.4	0	37.3	53.2	16.7	119.9	150.4	52.1	141.5	82.7	141.8	50.1	33.9	52.7
13. Calcium	mg·L⁻¹	*	2.32	0.74	2.04	3.71	1.23	3.69	5.71	3.49	4.14	4.12	1.21	4.17	2.78	0.65	2.68
14. Magnesium	mg·L⁻¹	*	0.56	0.09	0.54	0.74	0.21	0.71	0.88	0.32	0.82	1.03	0.38	1.02	0.61	0.09	0.60
15. Potassium	mg·L⁻¹	*	0.49	0.10	0.45	0.55	0.16	0.50	0.71	0.27	0.64	0.62	0.23	0.56	0.53	0.15	0.54
16. Sodium	mg·L⁻¹	*	0.79	0.13	0.76	0.94	0.18	0.96	0.91	0.09	0.87	1.04	0.26	1.07	0.72	0.11	0.71
17. Sulphate	mg·L⁻¹	*	10.7	1.47	11.4	11.1	2.21	10.6	10.8	0.97	10.7	7.4	1.04	7.4	6.9	0.98	6.9
18. Silica	mg·L⁻¹		3.19	1.05	3.08	2.97	1.21	3.05	2.05	1.08	1.87	3.63	1.37	3.71	2.56	1.12	2.54
19. Chloride	mg·L⁻¹		0.35	0.15	0.32	0.41	0.16	0.38	0.44	0.09	0.42	0.69	0.15	0.67	0.57	0.15	0.53
20. Nickel	μg·L⁻¹		13.07	5.91	13.1	8.05	4.51	7.5	9.71	6.35	7.0	0.12	0.24	0.1	0.05	0.05	0
21. Manganese	μg·L⁻¹	*	104.8	59.4	90.1	47.9	32.6	51.3	69.5	98.8	38.6	13.2	11.3	9.7	14.0	9.3	11.9
22. Aluminum	μg·L⁻¹	*	396.9	201.3	403.5	189.3	131.9	154.2	152.6	95.2	148.0	115.6	64.2	109.0	149.8	51.9	151.9
23. Copper	μg·L⁻¹	*	1.95	1.18	2.15	5.42	16.19	1.95	1.73	0.98	1.80	0.97	0.73	1.1	0.67	0.93	0.1
24. Zinc	μg·L⁻¹	*	10.26	5.81	10.10	7.33	4.99	6.90	6.15	3.59	6.00	5.49	2.95	4.90	7.60	7.39	5.35
25. Iron	μg·L⁻¹	*	142.1	117.2	101.6	114.9	96.9	87.0	94.5	67.9	73.0	113.9	81.8	87.0	83.2	51.5	77.6
26. Total organic carbon	mg·L⁻¹	*	3.66	1.94	3.70	4.86	2.51	4.35	6.15	3.41	6.30	8.75	2.82	9.10	8.27	2.01	8.50
27. Total Kjeldhal nitrogen	μg·L⁻¹	*	237	119	210	195	254	205	320	188	250	293	78	290	256	0.55	245
28. NH_3	μg·L⁻¹	*	21.33	11.79	19.05	19.58	10.62	16.30	22.87	8.75	18.50	22.75	7.39	21.90	20.91	7.59	19.00
29. $NO_2 + NO_3$	μg·L⁻¹	*	11.81	16.51	4.64	11.44	17.66	3.51	10.49	12.11	7.09	60.78	75.57	31.3	17.59	16.35	11.35
30. Total phosphorus	μg·L⁻¹		5.77	3.04	5.21	4.88	2.62	4.53	3.59	2.32	2.42	2.73	2.15	2.96	3.47	2.48	3.33
31. Soluble reactive phosphorus	μg·L⁻¹	*	1.14	0.46	1.09	1.41	0.66	1.29	1.47	1.18	0.98	1.21	0.38	1.10	1.35	0.55	1.39

* Variables used in principal factor analysis.

physical and biological data, including fisheries and waterfowl surveys, were included. Data from 123 lakes (69 from Ranger Lake and 54 from Wanapitei) were used.

To reduce the data to the major axes which account for most of the variation between the descriptor variables, a factor analysis was performed. Procedure FACTOR (Statistical Analysis System Institute Inc. 1982) was employed, using the squared multiple correlations of each variable with all the other variables as the prior communality estimate. To aid in the interpretation of the factor loadings, the VARIMAX rotation method was used (Statistical Analysis System Institute Inc. 1982). This is an orthogonal rotation method which simplifies the columns by maximizing the variance of the squared loadings of each column.

The results of the factor analysis performed on the combined data set (N = 123) are summarized in Table 7*a*. The percentages of the individual factor variability explained by each descriptor are illustrated in Figure 9. Four principal factors, accounting for 85.6% of the total rotated variance, were generated. Each is described below:

(1) *Factor 1* — Described as the "acidity" axis, factor 1 explains 35.9% of the variance. The chemical variables most strongly associated with factor 1 are all linked to acid deposition from smelting activities. This is apparent from the high positive correlations for several metals (Mn, Al, Zn, Cu and Fe) and sulphate (SO_4^{2-}). The strong negative correlation of pH (−0.888) and bicarbonate (−0.793) on this factor is consistent with this interpretation.

(2) *Factor 2* — Described as the "ionic strength" axis, factor 2 explains 24.0% of the variance. The strength of mineral influence in these head-water lakes is reflected in the second factor axis. The content of Ca^{2+} + Mg^{2+}, Na^+ and K^+ load positively on factor 2, as does the associated measure of ionic strength (KCORR). This axis could also be considered a measure of mineral buffering capacity as indicated by positive correlations of pH (+0.355) and bicarbonate (+0.538), in addition to the cations mentioned above. Sulphate has a moderate positive influence (+0.406) on the axis but loads more strongly on factor 1.

(3) *Factor 3* — Described as the "eutrophication"[1] axis, factor 3 explains 14.3% of the variance and is defined by both chemical and morphometric parameters that together describe a gradient of trophic conditions. The concentration of organic constituents and nutrients in the water is reflected by positive loadings for TOC, PO_4, Fe and TKN and coincides with the moderately high positive loading for wetland ratio (WETRAT = +0.356), which indicates a high proportion of riparian habitat versus open water in the wetland, indicative of a more mature successional stage. Negative loadings for maximum depth (−0.536) and lake area (−0.429) were also associated with this axis.

[1]The term "eutrophication" is not being used in its proper limnological context, as defined by quantitative characteristics such as total phosphorus, total nitrogen, chlorophyll *a* and Secchi disc transparency (Vollenweider and Kerekes 1982). It does, nevertheless, refer to a gradient of trophic conditions (defined by chemical and physical constituents), ranging from ultra-oligotrophic to oligotrophic and represents actual phases in the maturation of water bodies on the Precambrian Shield.

Table 7a
Principal factor analysis output for 123 lakes in the Ranger Lake and Wanapitei study areas*

Variable		Factor 1 "acidity"	Factor 2 "ionic strength"	Factor 3 "eutro-phication"	Factor 4 "morpho-metry"	Total
		Rotated factor pattern				
	Eigenvalue:	4.879	3.262	1.946	1.547	11.634
	% variance:	35.9%	24.0%	14.3%	11.4%	85.6%
1. pH		-.888	+.355			
2. Mn		+.879				
3. HCO_3		-.793	+.538			
4. Al		+.746	-.265	+		
5. SO_4		+.701	+.406			
6. Zn		+.579	-.284	+		
7. Cu		+.489	+.253			
8. NO_2+NO_3		-.459				
9. NH_3		-.269				
10. KCORR			+.946			
11. Ca + Mg		-.381	+.877			
12. Na			+.749			
13. K			+.343	+		
14. TOC		-.513		+.629	+	
15. PO_4				+.594		
16. Fe		+.342		+.548		
17. ZMAX		-.271		-.536		
18. WETRAT		+		+.356		
19. TKN		-		+.329		
20. DRAIN					+.732	
21. LA				-.429	+.696	
22. SDI					+.532	

* Factor scores are underlined if the correlation coefficient $r > 0.321$ ($p < 0.001$); otherwise $r > 0.254$ ($p < 0.01$). Positive or negative signs alone denote $r > 0.194$ ($p < 0.05$).

(4) *Factor 4* — Described as the "morphometry" axis, factor 4 explains 11.4% of the variance and is defined solely by positive loadings of physical variables, including lake area (LA), lake shape (SDI) and drainage basin area (DRAIN).

Using the procedures described above, separate factor analyses were performed on Wanapitei and Ranger Lake study lakes. The results of the combined analysis (Table 7*a*) were similar to those obtained for Wanapitei lakes only (Table 7*b*). Nearly one-half of the total variance was explained by factor 1, which was weighted heavily by parameters linked to acid deposition, including pH, Mn, Al and Zn. Factors 2, 3 and 4 contributed almost equally to the overall factor pattern generated comprising 41.9% of the variation. Factors 2 and 3 corresponded closely to the eutrophication and ionic strength factors defined in the combined analysis, whereas factor 4 was again weighted primarily by morphometric parameters.

The Ranger Lake factor output (Table 7*c*) was also similar to the combined scores. The combined factors 1 and 2 explained 58.1% of the 80.3% total rotated variance; however, parameters such as pH, Mn, Al and Zn tended to load onto both factor 1, ionic strength, and factor 2, eutrophication. Factors 3 and 4 explained only 22.2% of the variance. Although factor 3 includes several morphometric parameters, it is difficult to interpret factor 4. While the acidity gradient at Ranger Lake was less well defined than at Wanapitei, other chemical and physical relationships were similar.

Table 7b
Principal factor analysis output for 54 lakes in the Wanapitei study area*

Variable		Factor 1 "acidity"	Factor 2 "ionic strength"	Factor 3 "eutro-phication"	Factor 4 "morpho-metry"	Total
		Rotated factor pattern				
	Eigenvalue:	5.444	2.187	2.172	2.051	11.854
	% variance:	35.7%	14.3%	14.2%	13.4%	77.7%
1. pH		+.916		+		
2. Mn		-.847				
3. HCO_3		+.865		+.359		
4. Al		-.849				
5. SO_4				+.541		
6. Zn		-.834				
7. Cu		-	+.543			
8. NO_2 + NO_3					-.445	
9. NH_3					-.391	
10. KCORR		+.548		+.702		
11. Ca + Mg		+.728		+.600		
12. Na		+.364		+.397		
13. K			+.358			
14. TOC		+.443	+.748			
15. PO_4			+.632			
16. Fe		-	+.377	-.588		
17. ZMAX		+	-.445			
18. WETRAT			+.377		-	
19. TKN						
20. DRAIN					+.729	
21. LA					+.785	
22. SDI					+.506	

* Factor scores are underlined if the correlation coefficient $r > 0.443$ ($p < 0.001$); otherwise $r > 0.354$ ($p < 0.001$). Positive or negative signs alone denote $r > 0.273$ ($p < 0.05$).

The results obtained in the present study are similar to those obtained by Pitblado *et al.* (1980) following an analysis of water chemistry data (23 variables) collected for 187 lakes within 200 km of Sudbury. A principal component analysis showed that most of the chemical variability in these systems was attributable to four components defined as nutrient status, buffering status, atmospheric deposition status and sodium chloride status. In our study, ionic composition, nutrient levels and morphometric parameters combined to describe a major portion of the head-water lake environments measured. Because our waterfowl studies were designed to assess the effects of aquatic ecosystem acidification on a cross-section of head-water lakes in northeastern Ontario, further analyses of habitat correlates were performed using scores from the factor analysis performed on all study lakes combined (Table 7*a*).

2.2. Geochemistry and bedrock sensitivity

To assess the importance of bedrock sensitivity on the geochemistry of study lakes, a one-way analysis of variance (ANOVA) was performed on factor scores derived from the combined factor analysis for lakes grouped by the five bedrock sensitivity classifications defined in Appendix 1. Significant paired comparisons between bedrock sensitivity groups were determined using Tukey's Studentized range test. The results are depicted in Figure 10. Geological influence was substantial in the distribution of scores on both factor 1, acidity, and factor 2, ionic strength. No significant differences among bedrock types were found on factor 3, eutrophication, and only minor differences were apparent on factor 4, morphometry.

In general, the physiographic features of lakes and watersheds were similar between study areas and among bedrock sensitivity groups (Table 6). These lakes can be described simply as small, shallow head-water lakes with moderately irregular shorelines. Most lakes drain relatively small watersheds, and often contain a well-developed littoral zone community, with considerable riparian habitat composed of ericaceous shrub and graminoid vegetation. The minor differences in lake morphometry between high-sensitivity lakes at Wanapitei (WHS) and both VIXEN and GONG lakes (Fig. 10) can be largely attributed to the rather confined catchment basins and smaller average size of WHS lakes compared with those sampled at Ranger Lake.

Those chemical and physical parameters that describe factor 3, eutrophication, showed no significant differences among bedrock types (Fig. 10). There is little evidence that bedrock sensitivity explains any of the variability in organic carbon content (TOC), nutrient levels (PO_4 and TKN) or lake morphometry (ZMAX, WETRAT and LA). Although arbitrary in nature, the fixed boundary system for trophic categories proposed by Vollenweider and Kerekes (1982) does provide a clear definition of trophic terminology. The majority of lakes sampled in our study (64 and 43% at Ranger Lake and Wanapitei, respectively) would be classed as ultra-oligotrophic (TP $\leq$ 4.0 $\mu g \cdot L^{-1}$), with the remaining lakes categorized as oligotrophic (TP $\leq$ 10.0 $\mu g \cdot L^{-1}$) (Table 8). Fewer than 3% (N = 3) of the lakes sampled could be considered mesotrophic (TP = 10–35 $\mu g \cdot L^{-1}$). In our study, median levels of total phosphorus (TP) ranged from 2.4 $\mu g \cdot L^{-1}$ (WLS) to 5.2 $\mu g \cdot L^{-1}$ (WMS) (Table 8), and are, on average, lower than levels found in dilute soft-water lakes elsewhere in the Precambrian Shield (ca. 7 $\mu g \cdot L^{-1}$ at the Experimental Lakes Area) (Armstrong and Schindler 1971). Average total nitrogen (TN = TKN + (NO_2 + NO_3) + NH_3) levels ranged from a low of 224.8 $\mu g \cdot L^{-1}$ (WMS) to 343.2 $\mu g \cdot L^{-1}$ (VIXEN). Phosphorus is clearly the element in short supply in these dilute head-water lakes. The ratio of total nitrogen (TN) to total phosphorus (TP) was particularly high for the least sensitive bedrock groups in either study area (VIXEN (118:1) and WLS (115:1) lakes), compared with the moderate (WMS = 50:1) and high (WHS = 45:1) sensitivity systems at Wanapitei (Table 8).

Pitblado *et al.* (1980) found that nutrient status was extremely important in characterizing lakes in the greater Sudbury area and suggested that it reflected the overriding influence of nutrient abundance in lentic systems. However, the productive nature of many lakes sampled was directly related to cultural eutrophication, as many lakes contained considerable shoreline (cottage) development. Although Yan and Miller (1984) detected remarkably high average NO_2 + NO_3 levels in acidic lakes close to Sudbury (Table 8), Pitblado *et al.* (1980) found no relationship between nutrient

Table 7c
Principal factor analysis output for 69 lakes in the Ranger Lake study area*

Variable		Factor 1 "acidity"	Factor 2 "ionic strength"	Factor 3 "eutro-phication"	Factor 4 "morpho-metry"	Total
		Rotated factor pattern				
	Eigenvalue:	5.386	3.682	1.759	1.701	12.528
	% variance:	34.5%	23.6%	11.3%	10.9%	80.3%
1. pH		+.816	-.400			
2. Mn		-.407	+.514	+		
3. HCO_3		+.893	-			
4. Al		-.470	+.741			
5. SO_4		+	+.585			
6. Zn		-.429	+.379		+.401	
7. Cu		+			+.580	
8. NO_2+NO_3		+.462		+.439	+	
9. NH_3					+.652	
10. KCORR		+.959				
11. Ca + Mg		+.946				
12. Na		+.875				
13. K		+		-.419	+.367	
14. TOC			+.664			
15. PO_4		-	+.574	-		
16. Fe			+.755			
17. ZMAX			-.606	+	+.377	
18. WETRAT					-.496	
19. TKN			+.377			
20. DRAIN		+		+.664		
21. LA		-	-.392	+.609		
22. SDI				+.446		

* Factor scores are underlined if the correlation coefficient $r > 0.379$ ($p < 0.001$); otherwise $r > 0.302$ ($p < 0.01$). Positive or negative signs alone denote $r > 0.232$ ($p < 0.05$).

Figure 10
Comparisons among bedrock sensitivity groups (Ranger Lake—VIXEN and GONG; Wanapitei—high (WHS), moderate (WMS) and low (WLS) sensitivity) for each of the four factors, illustrated by the 95% confidence intervals about the average factor scores. Significant differences between bedrock types (using Tukey's multiple-range test) are also depicted

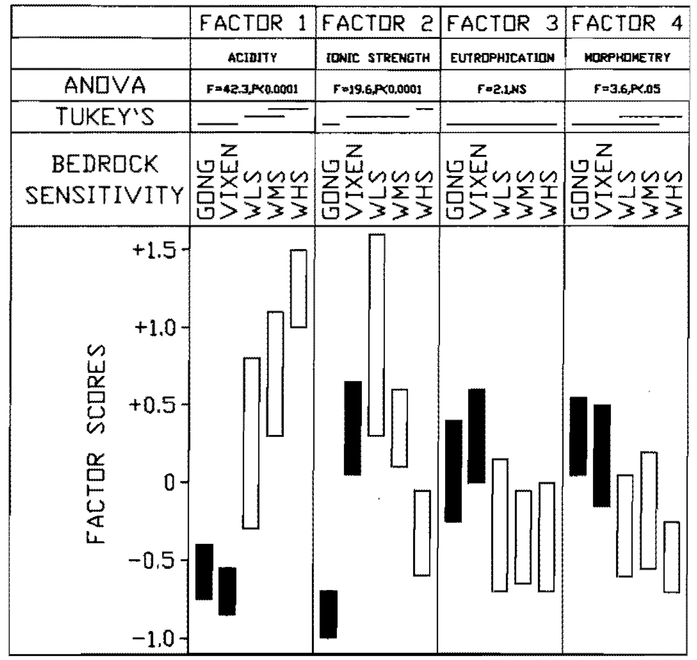


Table 8
Comparison of average nutrient levels (μg·L^{-1}) and nutrient ratios in study lakes, including the Sudbury Environmental Study (SES) lakes (modified from Yan and Miller 1984), according to bedrock sensitivity groups (Ranger Lake—VIXEN and GONG; Wanapitei—high (WHS), moderate (WMS) and low (WLS) sensitivity)*

Location	N	Distance from Sudbury, km	Parameter, μg·L^{-1}*† TP	TN	TKN	NO_2+NO_3	NH_3	Ratio TIN/TN	TN/TP
SES									
Hannah		4	5.7	743.3	150	576.0	17.3	0.798	130.4
Middle		5	7.3	660.0	180	442.0	38.0	0.727	90.4
Lohi		11	6.1	338.5	193	76.9	68.6	0.429	55.5
Clearwater		13	6.5	199.8	102	61.6	36.2	0.489	30.7
Labelle		27	10.7	300.9	256	20.3	24.6	0.149	28.1
Nelson		28	4.6	218.2	178	21.3	18.9	0.184	47.4
Mountaintop		52	6.4	251.6	191	36.5	24.1	0.241	39.3
Wanapitei									
WHS	20	56	5.8 (5.2)	270.1 (233.7)	237 (210)	11.8 (4.6)	21.3 (19.1)	0.123 (0.101)	46.6 (44.9)
WMS	24	58	4.9 (4.5)	226.0 (224.8)	195 (205)	11.4 (3.5)	19.6 (16.3)	0.137 (0.088)	46.1 (49.9)
WLS	11	44	3.6 (2.4)	353.4 (275.6)	320 (250)	10.5 (7.1)	22.9 (18.5)	0.095 (0.093)	98.2 (114.8)
Ranger Lake									
GONG	34	210	3.5 (3.3)	294.5 (275.4)	256 (245)	17.6 (11.4)	20.9 (19.0)	0.131 (0.110)	84.1 (83.5)
VIXEN	35	230	2.7 (2.9)	376.6 (343.2)	293 (290)	60.8 (31.3)	22.8 (21.9)	0.222 (0.155)	139.5 (118.3)

* Median values are expressed in parentheses.
† Abbreviations used: TP = total phosphorus; TN = [TKN + NH_3 + (NO_2+NO_3)]; TKN = total Kjeldhal nitrogen; TIN = [NH_3+ (NO_2+NO_3)].

status and proximity to Sudbury. In our study, concentrations of nitrogen and phosphorus were unrelated to smelter influence and were characteristic of most dilute, oligotrophic lakes on the Precambrian Shield. Variations in nutrient levels reflect the influence of morphometric and lithological factors or natural loadings of nutrients (particularly phosphorus) to these lakes.

Although physically similar, the ionic and trace metal composition of study lakes varied dramatically between study areas and among bedrock sensitivity groups. Major differences in surface water acidity were found between the two study areas (Fig. 11). Whereas a broad range of lake pH occurred across both areas, more than 66% of the lakes sampled at Wanapitei were acidic (pH < 5.5), compared with less than 10% at Ranger Lake. The results of the factor analysis suggest that a gradient of acidification exists between the two areas, ranging from heavily stressed and acidic lakes situated on moderately to highly sensitive bedrock at Wanapitei (WMS and WHS) to less heavily affected lakes located on less sensitive bedrock in both study areas (WLS, VIXEN and GONG) (Fig. 10). In the greater Sudbury area, Pitblado *et al.* (1980) found that both buffering status and atmospheric deposition status of lakes were related to the impact of atmospherically conveyed contaminants from the Sudbury smelting complex, as well as to the moderating influences of bedrock lithology. Brousseau *et al.* (1985) examined the role of bedrock and surficial geology in determining the sensitivity to acidification of lakes in the Thunder Bay area. They found that, in an area receiving low to moderate acid deposition, the small head-water lakes with low drainage basin/lake area ratios were extremely sensitive to acidification (alkalinity < 40 μeq·L^{-1}).

To show the degree of acidification of our study lakes, standard empirical relationships were employed using non-marine adjusted ion balance data. Comparisons of the chemical composition of fresh waters in unpolluted and polluted areas of Scandinavia and eastern North America are presented in Appendix 7. The ionic composition of lake water changes in response to acid deposition. The alteration

Figure 11
Comparison between lake pH distributions in the Ranger Lake and Wanapitei study areas

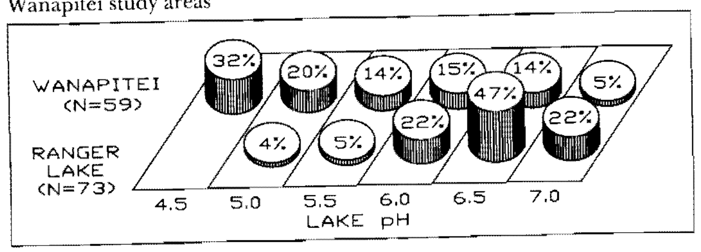
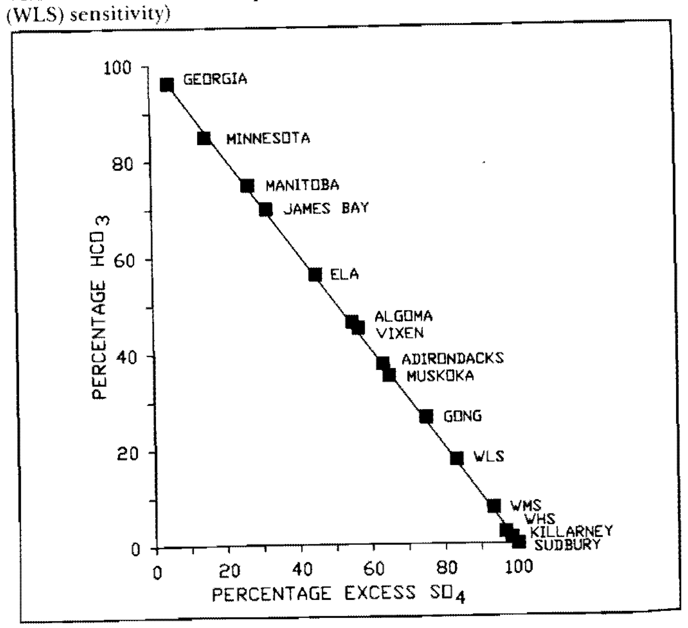


Figure 12.
The linear relationship between the percentage composition of bicarbonate (HCO_3^-) versus excess sulphate (SO_4^{2-}) anions in polluted and unpolluted fresh waters in eastern North America and comparisons among bedrock sensitivity groups in the present study (Ranger Lake—VIXEN and GONG; Wanapitei—high (WHS), moderate (WMS) and low (WLS) sensitivity)



is usually manifested as decreased acid-neutralizing capacity (i.e., alkalinity, HCO_3^-) and increased excess sulphate (SO_4^{2-}) concentrations; little, if any, change is detected in the relative concentrations of cations (Henriksen 1982).

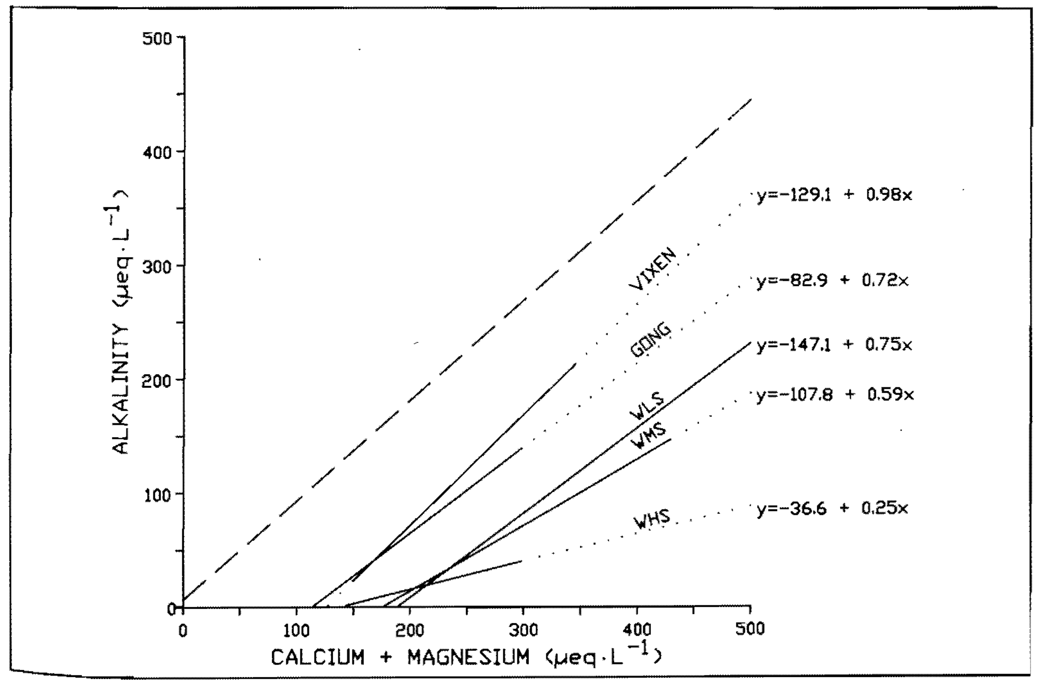
Given the predictable nature of these changes, it is possible to evaluate the influence of geography on acid deposition effects by comparing appropriate ion ratios. The alkalinity to excess sulphate (Alk:SO_4^{2-}) ratio provides a direct indication of the sulphate for bicarbonate replacement that occurs during the acidification process. It will range from large values (>10) when buffering is abundant (i.e., in relatively insensitive terrain) and SO_4^{2-} is low (i.e., low deposition areas) to values of zero when alkalinity is zero. In a recent evaluation of the regional acidification of lakes in eastern Canada using ion ratios, Jeffries (1986) illustrated that a large area of south-central Ontario (Muskoka–Haliburton) and southern Quebec (Laurentians) has been acidified, with the most dramatic evidence of acidification found in smaller areas of Nova Scotia and New Brunswick where terrain sensitivity is high and deposition is moderate. In unpolluted areas, the proportion of HCO_3^- in lake water exceeds SO_4^{2-}, as is the case in the Experimental Lakes Area of north-western Ontario (Beamish *et al.* 1976) and elsewhere in eastern Canada (Fig. 12 and App. 7).

In the present study, excess sulphate (SO_4^{2-}) concentrations are uniformly high at Wanapitei (median = 10.9 mg·L^{-1}, range 7.85–16.3), with correspondingly low Alk:SO_4^{2-} ratios exhibited. Similar ion balance data have been obtained from other Sudbury locations (Yan and Miller 1984) and Killarney (Beamish and Harvey 1972) (Fig. 12). Excess SO_4^{2-} concentrations were, on average, lower at Ranger Lake (median = 7.27 mg·L^{-1}, range 4.58–9.66), and overlapped Alk:SO_4^{2-} balances for lakes in south-central Ontario (Muskoka–Haliburton) (Dillon *et al.* 1980), in the Laurentian Mountain region of Quebec (Rodrigue and DesGranges 1986) and the Adirondack Mountain region of the north-eastern U.S.A. (Charles 1985).

The Alk:SO_4^{2-} ratio also differed dramatically among bedrock sensitivity groups in the present study (Fig. 12). These results may be attributed to the combined influence on water quality of the high deposition from the point source in Sudbury and/or high terrain sensitivity. At Wanapitei, the average Alk:SO_4^{2-} ratios decline from low (WLS = 0.24) to moderate (WMS = 0.08) to high (WHS = 0.02) bedrock sensitivities. This result is not a function of local variability in sulphate inputs, because no obvious correlation was found between smelter distance and either lake pH or bedrock types (Fig. 5). Low-sensitivity lakes (WLS) are clustered between 38 and 57 km of the smelter (mean = 46 km), and yet are characterized by relatively high pHs (median = 6.33), whereas the most sensitive lakes (WHS) (median pH = 4.65) were on average 55 km (range 43–64 km) from the smelter. These results show that lake buffering capacity in an acid-stressed area was related to the bedrock and surficial geology of the watershed, as was found in lakes sampled in the less heavily impacted Thunder Bay area (Brousseau *et al.* 1985). Despite uniformly high lake SO_4^{2-} levels at Wanapitei, bedrock geochemistry clearly sustains relatively normal major ion chemistry in certain lakes. All lakes with calcareous material in their watersheds (WLS; see App. 1) had elevated alkalinities, regardless of surficial geology. At Ranger Lake, SO_4^{2-} concentrations were nearly balanced by HCO_3^- in the moderately well-buffered VIXEN area (0.92), but were substantially lower in the moderately sensitive GONG area (0.37) (Fig. 12). Glacial lacustrine and outwash deposits found in the VIXEN area were associated with higher alkalinities on sensitive bedrock terrain, whereas glacial ground and end moraines contributed less to the acid-neutralizing capacity of lakes in the GONG area.

In our study, all lakes exhibited a bicarbonate (HCO_3^-) deficiency, as determined by the alkalinity versus Ca^{2+} + Mg^{2+} relationship (Alk : Ca^{2+} + Mg^{2+}) (Henriksen 1982) (Fig. 13). In systems unaffected by acid precipitation,

Figure 13.
Least squares regression between alkalinity (HCO_3^-) and the sum of calcium and magnesium (Ca^{2+} + Mg^{2+}) content (μeq·L^{-1}) for bedrock sensitivity groups (Ranger Lake–VIXEN and GONG; Wanapitei–high (WHS), moderate (WMS) and low (WLS) sensitivity). The "unpolluted dashed line" gives the pre-acidification relationship



the Ca and Mg content should balance the bicarbonate component, because alkalinity is derived from the weathering of calcareous materials that comprise the carbonate–bicarbonate buffer system (Henriksen 1982). Hence, the Alk : Ca^{2+} + Mg^{2+} ratio responds in a predictable manner to differing SO_4^{2-} levels, with the magnitude of the response (i.e., the ratio sensitivity) directly reflecting the terrain's capability of supplying basic cations and bicarbonate (Jeffries 1986). The Alk : Ca^{2+} + Mg^{2+} ratio will decrease from a value near unity (for the ideal clear lake experiencing low deposition) to zero as the alkalinity is exhausted, to negative values as the lake becomes increasingly acidic. In Figure 13, the extent of acidification is expressed by the distance below the "unpolluted" line where Alk = 0.91 (Ca^{2+} + Mg^{2+}). Of the 124 lakes sampled, 36% were located on terrain dominated by silicate rocks with a thin glacial overburden (GONG and WHS) and were, without exception, poorly buffered (App. 1). The remaining lakes showed a much broader range in buffering capacity attributable to the presence of thick calcareous overburden (VIXEN) or to either non-granitic (WLS) or conglomerate (WMS) underlying bedrock. At Wanapitei, lakes situated on highly sensitive bedrock (WHS) displayed almost no acid-neutralizing capacity whatsoever (slope = 0.25, R = 0.85, $p < 0.001$), a consequence of the naturally low buffering potential of the underlying bedrock having been completely exhausted by high SO_4^{2-} inputs over a number of decades. Other Wanapitei area lakes contained moderate (WMS: slope = 0.59, R = 0.81, $p < 0.001$) to substantial (WLS: slope = 0.75, R = 0.98, $p < 0.001$) buffering capacities. At Ranger Lake, the acid-neutralizing capacity was higher for both GONG (slope = 0.72, R = 0.78, $p < 0.001$) and particularly VIXEN (slope = 0.98, R = 0.93, $p < 0.001$) lakes.

The concentration of major cations (excluding H^+) decreased in the order $Ca^{2+} > Mg^{2+} > Na^+ > K^+$ in both areas. The relative proportions of major cations, expressed as the percentage of total cations (sum of Ca^{2+} + Mg^{2+} + Na^+ + K^+ + H^+) in Table 9, were consistent among bedrock sensitivity groups, with the exception that H^+ increased in relative importance in lakes situated on highly sensitive bedrock at Wanapitei (WHS). However, overall concentrations of cations, and in particular Ca^{2+} and Mg^{2+}, were much higher in WLS, WMS and VIXEN study lakes compared with WHS and GONG lakes (Table 9). The significant differences observed in factor 2, ionic strength (Fig. 10), reflect the enhanced buffering capacity present in WLS, WMS and VIXEN bedrock groups, compared with the rather dilute nature of GONG and WHS lakes. The geochemical similarity between GONG and WHS lakes, in particular, suggests that the potential for severe lake acidification exists with continued additions of sulphate to GONG lakes.

In instances of extreme acidification, the major ion content may be elevated (Yan and Miller 1984). Elevated levels of major cations (in particular Ca^{2+} and Mg^{2+}) have been observed in lakes near Sudbury that have been acidic for several decades, mainly because of increased rates of weathering in watersheds close to the point sources. Similarly, Yan and Miller (1984) attributed the decline in total anion levels in Sudbury Environmental Study lakes with increasing distance from Sudbury to elevated sulphate concentrations, arising primarily from the dry deposition of SO_2 in the immediate vicinity (within 20 km) of Sudbury. Elevated levels of major cations and anions were not observed in Wanapitei area lakes in our study; the levels were comparable with those of non-acidic lakes in south-central Ontario (Muskoka–Haliburton) and southern Quebec (Laurentians and Appalachians) (App. 7).

Trace metal concentrations also contributed to the significant trends observed on the acidity axis (Fig. 10). Concentrations of certain metals (including Cu, Ni, Zn and Fe) can be elevated in acid-stressed lakes because of increased atmospheric deposition (Beamish and VanLoon 1977; Jeffries and Snyder 1980). Acidification also increases the dissolution and mobilization of many elements—including Ag, Al, Cd, cobalt (Co), mercury (Hg), Mn, Ni, Pb and Zn—from soils in the watershed or from surface sediments of

Table 9
Comparison of average concentrations ($\mu eq \cdot L^{-1}$) and percentage composition of major ions (corrected for sea salts), according to bedrock sensitivity groups (Ranger Lake—VIXEN and GONG; Wanapitei—high (WHS), moderate (WMS) and low (WLS) sensitivity)

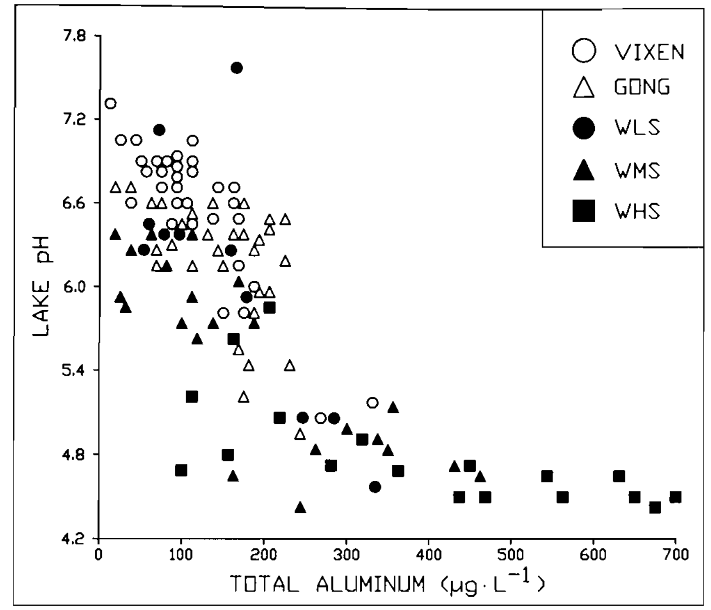
	Wanapitei												Ranger Lake							
	WLS (N = 11)				WMS (N = 24)				WHS (N = 20)				GONG (N = 34)				VIXEN (N = 32)			
	Mean		Median		Mean		Median		Mean		Median		Mean		Median		Mean		Median	
Major ions	$\mu eq \cdot L^{-1}$	%	$\mu eq \cdot L^{-1}$	%	$\mu eq \cdot L^{-1}$	%	$\mu eq \cdot L^{-1}$	%	$\mu eq \cdot L^{-1}$	%	$\mu eq \cdot L^{-1}$	%	$\mu eq \cdot L^{-1}$	%	$\mu eq \cdot L^{-1}$	%	$\mu eq \cdot L^{-1}$	%	$\mu eq \cdot L^{-1}$	%
Hydrogen	4.3	1.7	NA*	0.2	8.6	3.2	NA	0.8	23.8	11.3	NA	11.4	1.3	0.7	NA	0.3	0.8	0.5	NA	0.1
Calcium	284.6	66.2	206.1	66.9	184.8	61.2	183.8	61.1	115.5	51.3	101.3	50.3	138.0	63.1	132.9	62.3	195.2	62.4	197.5	62.4
Magnesium	70.4	18.3	65.6	18.7	58.8	19.9	55.4	18.5	43.8	19.9	42.3	19.3	46.8	21.7	45.4	21.9	73.7	23.7	76.0	23.3
Potassium	18.1	5.2	16.1	4.1	13.7	4.8	12.4	4.4	12.4	5.8	11.4	5.4	13.1	6.2	13.3	6.0	14.9	5.1	13.7	4.6
Sodium	29.2	8.6	27.8	7.6	31.0	10.7	32.5	10.1	25.9	11.8	26.0	11.2	17.8	8.3	18.3	7.9	26.9	8.4	28.5	8.5
Total cations	406.5		318.6		297.0		276.0		221.4		210.1		217.1		211.9		308.7		318.5	
Bicarbonate	119.8	25.8	52.0	19.0	37.3	11.5	16.7	7.9	3.4	1.1	0	0	50.0	24.3	52.6	26.0	132.2	41.9	141.0	45.7
Sulphate	224.1	74.1	221.1	80.9	229.9	88.4	217.9	91.8	221.1	98.8	236.2	99.9	142.8	75.6	141.7	73.7	154.1	57.8	153.7	53.7
Nitrate	0.2	0.1	1.0	0.1	0.2	0.1	1.0	0.3	0.2	0.1	1.0	0.1	0.3	0.1	1.0	0.3	0.9	0.3	1.0	0.6
Total anions	344.1		274.9		267.7		237.7		225.6		236.3		193.1		191.8		287.3		293.4	
	Mean		Median		Mean		Median		Mean		Median		Mean		Median		Mean		Median	
Water colour	25		20		25		20		14		10		25		20		44		38	
Organic Anion†	57.5		60.2		39.9		34.3		26.3		26.1		76.1		77.6		82.7		87.6	
Organic Anion‡	62.4		60.9		30.7		26.0		–4.2		–9.9		23.9		25.7		24.3		23.9	
Anions + Organic Anion[2]	401.6		321.1		307.5		279.5		251.0		246.4		269.3		261.3		373.5		385.6	

* NA = data not available.
† From Oliver *et al.* 1983.
‡ Organic Anion[1] = cations – anions.

acidic lakes (Wright and Gjessing 1976; Schindler *et al.* 1980; Campbell and Stokes 1985). Aluminum and Mn are particularly good indicators of watershed acidification because they are readily mobilized from soils by mineral acids and occur naturally at low levels in lakes on the Precambrian Shield (Scheider *et al.* 1979). In our study, both Al and Mn contributed to the gradient of acidity defined by factor 1. In both areas, Al and Mn were negatively correlated with pH (r_s = –0.72 and –0.58 at Ranger Lake and r_s = –0.81 and –0.81 at Wanapitei, respectively).

Acidification of fresh waters can influence metal–organism interactions in at least two ways: the decrease in pH may affect metal speciation in solution, or it may affect biological sensitivity at the level of the cell surface. In their review of acidification and toxicity of metals to aquatic biota, Campbell and Stokes (1985) concluded that a pH-dependent biological response has been documented over a realistic range of H^+ and metal concentrations for 6 of the 10 metals mentioned above (Al, Cd, Cu, Zn and, to a lesser extent, Hg and Pb). For those organisms that are inherently sensitive to pH values above 6, the pH–metal interactions are of little interest. However, for aquatic biota that can tolerate pH levels between 6 and 5, these interactions may be important, as this level of acidification (pH 6–5) corresponds to that at which a number of metals are mobilized (H^+–Al and H^+–Hg interactions). For example, lethal limits for Al have been set as low as 70 $\mu g \cdot L^{-1}$ for sensitive salmonids, although 200 $\mu g \cdot L^{-1}$ is the generally accepted level (Cronan and Schofield 1979). In lakes with pH greater than 6 in the present study, total aluminum levels rarely exceeded 200 $\mu g \cdot L^{-1}$ (Fig. 14). Elevated Al levels were more pronounced at Wanapitei where the proportion of low pH/high Al levels (>200 $\mu g \cdot L^{-1}$) followed the general gradient of lake acidity from low (WLS = 27%) to moderate (WMS = 38%) to high (WHS = 80%) bedrock sensitivities. A similar pattern also existed for Mn levels, with average

Figure 14
Relationship between lake pH and total aluminum levels ($\mu g \cdot L^{-1}$) by bedrock sensitivity groups (Ranger Lake—VIXEN and GONG; Wanapitei—high (WHS), moderate (WMS) and low (WLS) sensitivity)



levels at Ranger Lake (median = 10.7 $\mu g \cdot L^{-1}$) substantially lower than at Wanapitei (median = 59.5 $\mu g \cdot L^{-1}$), and well below average estimates for two non-acidic lakes (Labelle and Nelson) near Sudbury and assorted Muskoka–Haliburton lakes (Table 10).

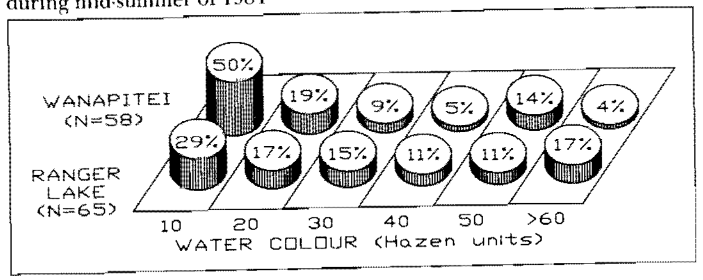
Yan and Miller (1984) reported that levels of Zn, Cu and Ni were elevated in acidic Sudbury Environmental Study lakes and concluded that this was a result of direct deposition. With the exception of Fe, which is a major lithological element, levels of metals in their study lakes

Table 10
Comparison of average trace metal levels ($\mu g \cdot L^{-1}$) in study lakes*

Location	N	Distance from Sudbury, km	Cu	Ni	Zn	Fe	Al	Mn
			Parameter, $\mu g \cdot L^{-1}$					
SES								
Hannah		4	1108.0	1865.0	120.0	47.0	1097.0	340.0
Middle		5	496.0	1068.0	91.3	143.0	–	354.0
Lohi		11	83.6	254.0	41.6	90.0	–	284.0
Clearwater		13	81.3	278.0	39.2	88.2	381.0	290.0
Labelle		27	4.8	12.7	10.6	73.0	73.3	74.8
Nelson		28	22.0	17.1	18.3	60.8	86.6	63.1
Mountaintop		52	24.0	12.2	28.0	215.0	440.0	120.0
Wanapitei								
WHS	20	56	1.9	13.1	10.3	142.1	396.9	104.8
			(2.2)	(13.1)	(10.1)	(101.6)	(403.5)	(90.1)
WMS	24	58	5.4	8.1	7.3	114.9	189.3	47.9
			(1.9)	(7.5)	(6.9)	(87.0)	(154.2)	(51.3)
WLS	11	44	1.7	9.7	6.2	94.5	152.6	69.5
			(1.8)	(7.0)	(6.0)	(73.0)	(148.0)	(38.6)
Ranger Lake								
GONG	34	210	<1	<1	7.6	83.2	149.8	14.0
					(5.4)	(77.6)	(151.9)	(11.9)
VIXEN	35	230	<1	<1	5.5	113.9	115.6	13.2
					(4.9)	(87.0)	(109.0)	(9.7)
Muskoka–Haliburton	15	214	<2	<2	5.0	98.0	42.0	33.0
					(3.8)	(20–230)	(8–91)	(15–71)

* Data from the Sudbury Environmental Study (SES) lakes are averages from 1976–78 data (modified from Yan and Miller 1984). Data from the present study are presented according to bedrock sensitivity groups (Ranger Lake—VIXEN and GONG; Wanapitei—high (WHS), moderate (WMS) and low (WLS) sensitivity). Median values are expressed in parentheses. Muskoka-Haliburton data (Dillon *et al.* 1980) are expressed as means with range in parentheses.

Figure 15
Comparison between the distributions of water colour (Hazen units) in littoral samples collected in the Ranger Lake and Wanapitei study areas during mid-summer of 1984



decreased with increasing distance from Sudbury (Table 10). Acid deposition in lakes studied by Pitblado *et al.* (1980) was indicated by high concentrations of SO_4^{2-}, Cu, Ni, Zn and Pb, with lakes north of Lake Wanapitei showing a noticeable influence of atmospheric deposition compared with lakes farther away from Sudbury. In our study, levels of Cu and Zn were only marginally higher in low- and moderate-sensitivity lakes at Wanapitei compared with either Ranger Lake or Muskoka–Haliburton lakes (Table 10). Nickel levels were uniformly higher at Wanapitei compared with Ranger Lake and might indicate the influence of direct deposition of this element.

In recent years, declines in trace metal emissions from Sudbury smelters have accompanied SO_2 emission reductions (Keller and Pitblado 1986). The reduced concentrations of Ni and Cu in many Sudbury area lakes may be of considerable biological importance, as alone, or in combination, high concentrations of these metals may be toxic to aquatic biota, particularly in association with low pH. In their comparison of average trace metal concentrations, Keller and Pitblado (1986) found that only a relatively small proportion of lakes (7 and 22%, respectively) had average Ni and Cu concentrations exceeding Ministry of Environment (1978) objectives (25 and 5 $\mu g{\cdot}L^{-1}$ for Ni and Cu, respectively) during the 1981–83 surveys compared with 1974–76 surveys (8 and 62% for Ni and Cu, respectively). Only 3 of the 55 lakes sampled at Wanapitei exceeded the 5 $\mu g{\cdot}L^{-1}$ level for Cu; no lakes sampled exceeded the 25 $\mu g{\cdot}L^{-1}$ level for Ni. Lead and Cd levels were barely detectable in surface waters sampled in the present study.

2.3. Organic acidity of surface waters

Mid-summer water samples collected in the littoral zone (< 1 m) of lakes in the present study ranged from clear to tea-coloured, with dark, humic waters occurring infrequently (Fig. 15). Average water colour estimates were substantially higher at Ranger Lake (median = 30 Hazen units) compared with Wanapitei (median = 18 Hazen units), with 17% of the lakes sampled at Ranger Lake noticeably coloured (≥55 Hazen units). At Wanapitei, 69% of the lakes sampled were clear (≤25 Hazen units) compared with only 46% at Ranger Lake. With the exception of VIXEN lakes (median = 38 Hazen units), median colour values were 20 Hazen units or less for all bedrock sensitivity groups (Table 9), with values ranging from 5 to 55 Hazen units. For both study areas, a good relationship exists between water colour and dissolved organic carbon content (TOC) (r_s = 0.87 and 0.77, $p<$ 0.0001, for Ranger Lake and Wanapitei, respectively). Organic carbon (TOC) content was a significant parameter in both factor 1, acidity, and factor 3, eutrophication (Fig. 9), and provides a good indication of aquatic ecosystem productivity. Within the range of head-water lakes sampled, clear, acidic waters were rather dilute in nature, and were more characteristic of ultra-oligotrophic conditions, whereas non-acidic, coloured waters often represented more productive wetlands, many of which bordered on mesotrophic conditions.

The toxicity of Al to aquatic biota may be greatly reduced through binding by organic acids (Baker and Schofield 1982). The ability of organic acids to bind Al and remove or greatly reduce its toxicity to biota could vary with the composition of the organic acidity, the levels of Al in the system and the pH of the system (ESSA 1986). It is unlikely that metal complexation by organic matter would severely limit potential toxicity of Al in our study lakes, given the rather low average levels of DOC recorded (Table 9).

Recent studies in Nova Scotia (Kerekes *et al.* 1984) and elsewhere (Gorham and Detenbeck 1986) have demonstrated problems associated with sulphate determinations in coloured, humic waters. These findings are particularly relevant in view of increased concerns about the relative importance of organic anions in the acidity of fresh-water systems receiving mineral acids in the form of airborne pollutants. Until recently, many studies investigating acidification effects in fresh waters have used the methyl thymol blue (MTB) method for sulphate (SO_4^{2-}) determination. Compared with ion chromatography (IC), the MTB method overestimates SO_4^{2-} in coloured waters (Cronan and Schofield 1979), thereby underestimating the contribution of organic anions to the dissociated hydrogen ion (H^+) concentrations of these surface waters.

In the present study, MTB SO_4^{2-} determinations were used in the calculation of charge balances (Table 9). To assess the analytical error introduced by MTB SO_4^{2-} determinations and evaluate the importance of organic acidity in our study lakes, charge balances were calculated by the Oliver *et al.* (1983) equation as follows:

$$\text{Organic Anion}^2 = K * Ct/(K + H^+) \qquad [2]$$

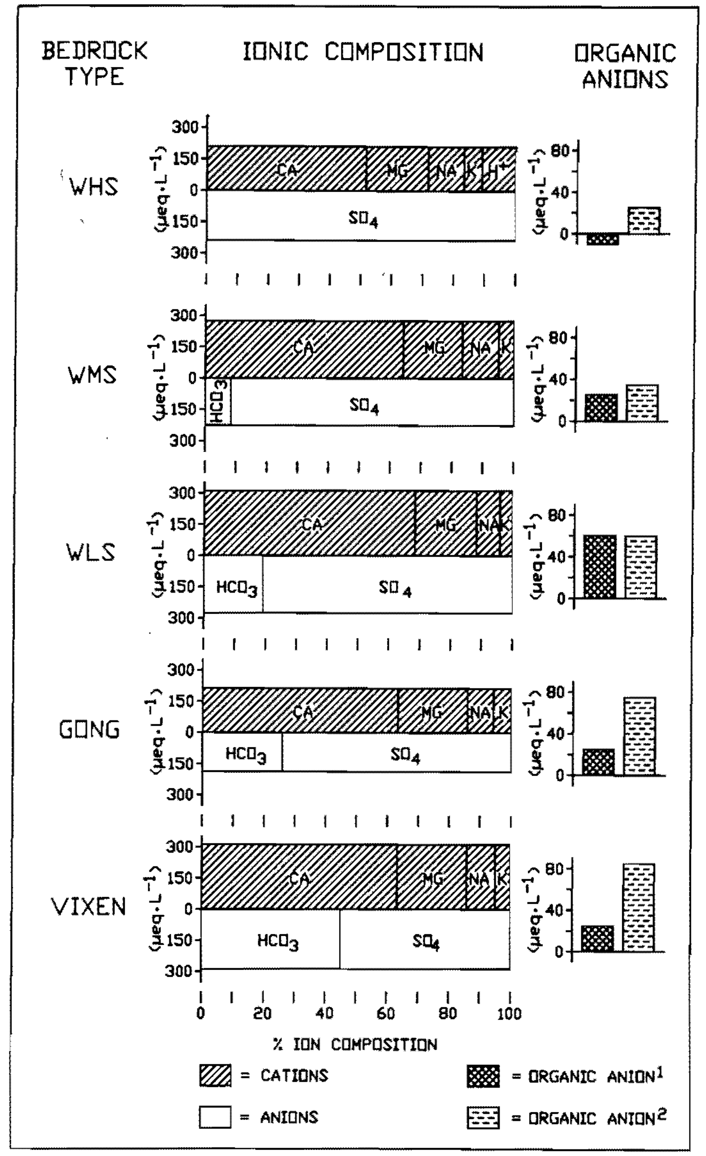
where Organic Anion² = organic anion concentration ($\mu eq{\cdot}L^{-1}$)
K = 10^{-pK}
pK = 0.958 + 0.90 pH − 0.039 $(pH)^2$
Ct = 10 × DOC = organic acid concentration ($\mu eq{\cdot}L^{-1}$)
DOC = dissolved organic carbon concentration ($mg{\cdot}L^{-1}$)

The traditional calculation of organic anion content using the deficit relationship (Organic Anion¹ = cations − anions (using MTB SO_4^{2-})) provides a reasonably good charge balance in the more dilute, clear-water systems at Wanapitei (WHS, WMS and WLS) (Fig. 16), where the combined influence of the SO_4^{2-} overestimation and organic anion content would be negligible. In the VIXEN and GONG areas, however, the deficit relationship (Organic Anion¹) explains only a fraction of the measurable Organic Anion² content. Overestimation of SO_4^{2-} levels would explain only a portion of the discrepancy between Organic Anion² and the deficit estimate (Organic Anion¹). However, the results of recent (May 1985) testing using duplicate measurements of both MTB and IC SO_4^{2-} determinations in 75 peatland water samples at Ranger Lake and Wanapitei (McNicol, unpubl.) indicate a good correlation (R^2 = 0.97) between the two estimates, with the following regression equation generated from the relationship:

$$SO_4^{2-}(IC) = \qquad [3]$$
$$0.09 + (0.914 \times SO_4^{2-}(MTB)) - (0.0118 \times \text{Colour})$$

where SO_4^{2-}(IC) and SO_4^{2-} (MTB) are expressed as $mg{\cdot}L^{-1}$, and Colour is expressed in Hazen units. Using mean values

Figure 16
The percentage composition and concentrations ($\mu eq{\cdot}L^{-1}$) of major ions by bedrock sensitivity groups (Ranger Lake—VIXEN and GONG; Wanapitei—high (WHS), moderate (WMS) and low (WLS) sensitivity), and comparisons between two estimates of organic anion concentrations ($\mu eq{\cdot}L^{-1}$)



for SO_4^{2-} and water colour from Table 9, the above equation predicts the following overestimates of sulphate ions using MTB: WHS = + 10.2%, WMS = + 11.6%, WLS = + 11.7%, GONG = + 13.0%, and VIXEN = + 16.8%. The overestimate using the MTB method, over the range of water colours present in head-water lake samples, would rarely exceed 17% of the IC measurement. Including other cations (Al^{3+}, Fe^{3+}, Mn^{2+} and NH_4^+) in the calculation of total cations would improve charge balances considerably at Ranger Lake (GONG—Organic Anion¹ = 45.3; VIXEN—Organic Anion¹ = 43.6), but the reverse would be true at Wanapitei (WLS = 86.5, WMS = 58.7 and WHS = 49.9).

Although the results are inconclusive, the data suggest that inaccuracies in charge balance calculations, arising from MTB SO_4^{2-} measurements, would be minimal over the range of water colour and organic carbon content in this study. Only in a small percentage of lakes studied, particularly at Ranger Lake, would organic anions constitute a major ionic constituent, but they would not contribute substantially to the acidity of these systems.

3. Biological response to lake acidification

3.1. Fish community structure

Much of the historical information on fish communities in Ontario deals specifically with the large game species. In the sensitive region between the Algonquin highlands and Sault Ste. Marie, lake trout (*Salvelinus namaycush*), brook trout (*Salvelinus fontinalis*) and centrarchid communities are common. However, the breakdown of fish communities by lake-size categories in this region confirmed that lakes with no fish or no game fish are mostly less than 10 ha in size (Minns 1981). Head-water lakes, which are most susceptible to acidification and include aquatic habitats most exploited by waterfowl (Fig. 8), are not necessarily critical habitat for most large game fish. However, trophic relationships in such systems are influenced by the occurrence, composition and abundance of forage or non-game fish species.

The structure of small non-game fish assemblages and the physical and chemical factors that determine their organization were examined in head-water lakes in the two study areas. Seven cyprinid species were commonly captured by our minnow traps. These species were northern redbelly dace (*Chrosomus eos*), finescale dace (*Chrosomus neogaeus*), pearl dace (*Semotilus margarita*), fathead minnow (*Pimephales promelas*), common shiner (*Notropis cornutus*), creek chub (*Semotilus atromaculatus*) and blacknose dace (*Rhinichthys atratulus*). Because populations consisting totally of *C. eos* × *C. neogaeus* hybrids were found, they and their parent species have been designated as *Chrosomus* spp. Non-cyprinids taken in minnow traps were white sucker (*Catostomus commersoni*), brook stickleback (*Culaea inconstans*), Iowa darter (*Etheostoma exile*), yellow perch (*Perca flavescens*), pumpkinseed sunfish (*Lepomis gibbosus*) and rock bass (*Ambloplites rupestris*). Species taken infrequently were brook trout (*Salvelinus fontinalis*), lake chub (*Couesius plumbeus*), blacknose shiner (*Notropis heterolepis*) and golden shiner (*Notemigonus crysoleucas*) at Ranger Lake, and smallmouth bass (*Micropterus dolomieui*), northern pike (*Esox lucius*) and brown bullhead (*Ictalurus nebulosus*) at Wanapitei. A complete summary of minnow trapping results is presented in Appendix 8.

Differences in fish occurrence and species composition were found between the two study areas (Table 11). The most striking difference was that 43% of lakes at Wanapitei had no fish compared with only 12% without fish at Ranger Lake. In the 31 lakes at Wanapitei that contained fish, the families represented were percids (68%), cyprinids (52%), catostomids (26%), centrarchids (23%) and gasterosteids (19%). The most prevalent species recorded was yellow perch, which was found in 45% of the lakes supporting fish. Of the cyprinids, only *Chrosomus* spp. were common (32%). At Ranger Lake, no centrarchids were recorded, and only one lake (002R) contained yellow perch (App. 8*a*). Cyprinids were nearly ubiquitous in their distribution, with white sucker and brook stickleback also common.

Data on the distribution of cyprinids at Ranger Lake have been analysed for evidence that their occurrence was related to the chemical and physical characteristics of the lakes (Bendell and McNicol 1987*b*). To summarize these findings, the six major cyprinid taxa can be readily divided into two ecological groups. The "pond-dwelling" species, such as *Chrosomus* spp., pearl dace and fathead minnow, often occur together in north-eastern Ontario in "stained, acid waters of beaver ponds and small lakes," often in association with brook stickleback (Scott and Crossman 1973). The other three species, common shiner, creek chub and blacknose dace, are common "stream-dwelling species"

Table 11
The number and percentage of lakes occupied by non-game fish species and families in the Ranger Lake and Wanapitei study areas

Species		Fish occurrence			
		Ranger Lake ($N = 69$)		Wanapitei ($N = 54$)	
Common name	Scientific name	No.	%	No.	%
F. Cyprinidae		61	88.4	16	29.6
1. *Chrosomus* spp.		46	66.7	10	18.5
2. common shiner	*Notropis cornutus*	22	31.9	3	5.6
3. pearl dace	*Semotilus margarita*	33	47.8	6	11.1
4. creek chub	*Semotilus atromaculatus*	27	39.1	1	1.9
5. fathead minnow	*Pimephales promelas*	20	28.9	6	11.1
6. blacknose dace	*Rhinichthys atratulus*	15	21.7	0	0
7. golden shiner	*Notemigonus crysoleucas*	2	2.9	4	7.4
F. Centrarchidae		0	0	7	12.9
1. rock bass	*Ambloplites rupestris*	0	0	3	5.6
2. pumpkinseed sunfish	*Lepomis gibbosus*	0	0	6	11.1
F. Percidae		7	10.1	21	38.9
1. yellow perch	*Perca flavescens*	1	1.5	14	25.9
2. Iowa darter	*Etheostoma exile*	6	8.7	7	12.9
F. Gasterostidae		25	36.2	6	11.1
1. brook stickleback	*Culaea inconstans*	25	36.2	6	11.1
F. Catostomidae		28	40.6	8	14.8
1. white sucker	*Catostomus commersoni*	28	40.6	8	14.8
Fish absent		8	11.6	23	42.6
Fish present		61	88.4	31	57.4

that use the gravel beds of streams as spawning sites. In contrast, typical pond-dwelling species, *Chrosomus* spp. and the fathead minnow, breed in ponds with soft organic substrates, whereas the pearl dace breeds either over hard substrates in streams (Langlois 1929) or "in the vegetation on the periphery of lakes" (Tallman *et al.* 1984). These pond species were widespread, and at least one of them occurred in all but 4 of the 61 lakes supporting cyprinids (App. 8*a*). In contrast, stream species were absent from 25 lakes, and appeared to constitute a distinct assemblage of frequently co-occurring species. Stream species occurred more often than expected with other stream species than without them (χ^2 test, Table 12). Fathead minnows and pearl dace did not occur more often in the presence of stream species, and *Chrosomus* spp. tended to occur more frequently in their absence. Although these chi-squared tests are not independent, they strongly support the hypothesis that the pond/stream dichotomy is a major structural feature of the cyprinid community at Ranger Lake. This analysis can be extended to the most commonly recorded non-cyprinid species as well. The brook stickleback, a pond-breeding species, co-occurred with stream cyprinids in only 10 of the 25 lakes where it was found ($\chi^2 = 0.20$, $p > 0.05$), whereas the white sucker, which spawns in the gravel beds of streams, occurred with stream cyprinids in 20 of the 27 lakes in which it was recorded ($\chi^2 = 6.08$, $p < 0.05$).

3.2. Fish community structure and lake acidification

Comparative tolerances of native fish species to lake acidity have been investigated extensively, usually through population surveys (Harvey 1975; Kelso *et al.* 1981; Harvey and Lee 1982; Somers and Harvey 1984; Beggs *et al.* 1985; IEC Beak 1985; Frenette *et al.* 1986), experimental acidification (Schindler and Turner 1982; Mills 1985; Mills *et al.* 1987; Schindler *et al.* 1985) or single-species toxicity or condition tests (Beamish 1972, 1974; Mount 1973; Ryan and Harvey 1980; Baker and Schofield 1982; Holtze 1983; Fraser

Table 12
Number of occurrences of each cyprinid species among 58 lakes in the Ranger Lake study area, proportion of those occurrences with (other) stream species, expected proportion if occurrences were random with respect to (other) stream species, and chi-squared tests of independence of occurrence of each species with (other) stream species (***$p<0.001$, **$p<0.01$, *$p<0.05$)

Species	Number of occurrences, N	Proportion of occurrences with (other) stream species	Expected proportion of occurrences with (other) stream species	Chi-squared statistic
Pond species				
Chrosomus spp.	44	0.48	0.57	6.27*
pearl dace	32	0.59	0.57	0.18
fathead minnow	20	0.65	0.57	0.82
Stream species				
common shiner	20	0.80	0.50	10.99***
creek chub	26	0.77	0.47	17.49***
blacknose dace	15	0.87	0.53	8.96**

and Harvey 1984; Lemly and Smith 1985). Most studies have concentrated on species that are important to sport or commercial fisheries, particularly the salmonidae (brook trout, lake trout, rainbow trout (*Salmo gairdneri*)) (Packer and Dunson 1972; Menendez 1976; Robinson *et al.* 1976; Trojnar 1977; MacCrimmon *et al.* 1983; Moreau and Barbeau 1983; Brown and Eales 1984; Neville 1985), rather than fish communities composed of small non-game species.

Relating changes in fish communities to acidification requires a knowledge of community composition prior to acid inputs. For many regions, however, fish data are sparse or available only for a few lakes (rarely head-waters) or selected species (rarely minnows). In Ontario, Minns (1981) confirmed that most small lakes between 10 and 100 ha located in watersheds 2C-E in the Sault Ste. Marie – Sudbury axis (Fig. 3*d*) would contain primarily brook trout communities. Non-game fish assemblages only would occur in most lakes less than 10 ha. In the present study, we assessed the biological, chemical and physical characteristics of small lakes subjected to varying atmospheric inputs and geochemical influences that together affect fish community composition.

Historical information on fish population status in lakes surrounding Sudbury is scarce. The extent to which trace metal toxicity, in association with low pH, can be implicated in observed fish population losses in Sudbury area lakes is not clear (Beamish and Harvey 1972; Ministry of Environment 1979). Assuming that metal emissions are roughly proportional to SO_2 emissions, Keller and Pitblado (1986) concluded that trace metal toxicity likely played a large role during the period when most fishery losses occurred, 1950–70 (Beamish and Harvey 1972; Gunn 1982), and when trace metal emissions were probably two to four times present levels. However, there are indications of improvements in the fishery in response to recent pH increases and metal concentration decreases in some Sudbury area lakes (Gunn and Keller 1984). Fewer lakes currently exceed Ministry of Environment (1978) objectives for protection of fish and aquatic life (1981–83 surveys) compared with 1974–76 (Keller and Pitblado 1986).

The results of non-game fish sampling conducted in the Wanapitei area support previous findings of fishery losses in the greater Sudbury area. Where lake pH exceeded 6, fish assemblages were dominated by diverse cyprinid populations in both areas (Fig. 17). A very simple non-cyprinid community, dominated by the acid-tolerant yellow perch, was commonly found between pH 6 and 5 at Wanapitei. Of the 30 acidic lakes (pH < 5.5) examined at Wanapitei, more than 66% were fishless. No differences in lake pH were found between cyprinid and fishless groups at Ranger Lake (t-test, $t = 2.31$, $p > 0.05$). Significant differences were apparent among fish assemblages at Wanapitei (ANOVA, $F = 21.82$, $p < 0.0001$); cyprinid lakes had a significantly higher average pH than either non-cyprinid or fishless lakes (Tukey's multiple-range test).

At Ranger Lake, the occurrence of eight fishless lakes was unrelated to lake pH. Nearly 10% of the 63 non-acidic lakes sampled were fishless, which is similar to the 13% of the non-acidic lakes ($N = 24$) found to be devoid of fish at Wanapitei. These results suggest that a small proportion of head-water pond and lake systems do not sustain fish populations of any kind, as a result of natural causes, including biogeographic isolation and/or winter anoxia (IEC Beak 1985).

The linear regression of species richness on pH was highly significant at Wanapitei ($R^2 = 0.65$, $Y = -5.24 + 1.22$ pH, $p < 0.0001$). This suggests that species richness decreases as acidification increases because of the loss of acid-intolerant species. Conditions at Ranger Lake favour a diverse cyprinid community, with no obvious relationship observed between species richness and lake pH ($R^2 = 0.11$, $Y = 0.06 + 0.47$ pH, $p > 0.05$).

Cyprinids were less tolerant of acidic conditions than non-cyprinids. A comparison among individual species distributions in the two study areas in relation to lake pH is given in Figure 18. The average distribution (mean ± 1 SD) of several cyprinids, including common shiner, fathead minnow, creek chub and blacknose dace, was above pH 6 in both areas, although the last two species were not recorded at Wanapitei. No cyprinids were common below pH 5.5, although the typical pond-dwelling species, *Chrosomus* spp. and pearl dace, were found in the pH 6–5.5 range, along with white suckers. Below pH 5.5, several non-cyprinid species, particularly brook stickleback, pumpkinseed sunfish, Iowa darter and yellow perch, were recorded. Below pH 5, lakes which contained fish were infrequent.

The loss of sport and commercial fish populations from acidified lakes and streams is usually detected when the pH of fresh waters is less than 5. Although recruitment failures are probably the ultimate cause for loss of fish populations, changes in food webs accelerate the process (Mills and Schindler 1986). As pH of an aquatic ecosystem

Figure 17
Range and 95% confidence intervals about the mean lake pH of major fish assemblage types (cyprinid, non-cyprinid and fishless lakes) in the Ranger Lake and Wanapitei study areas

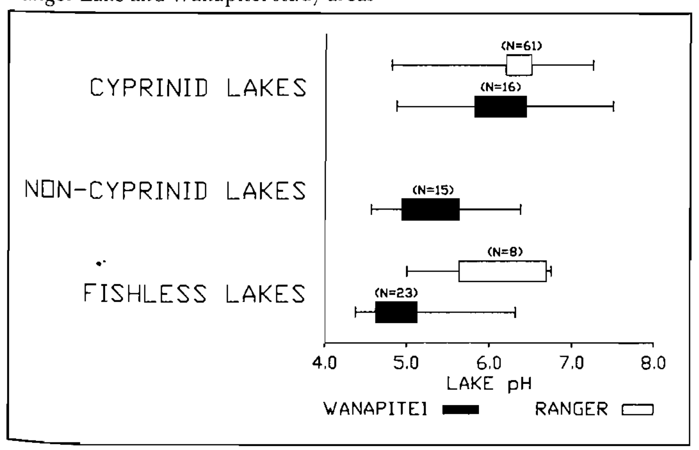
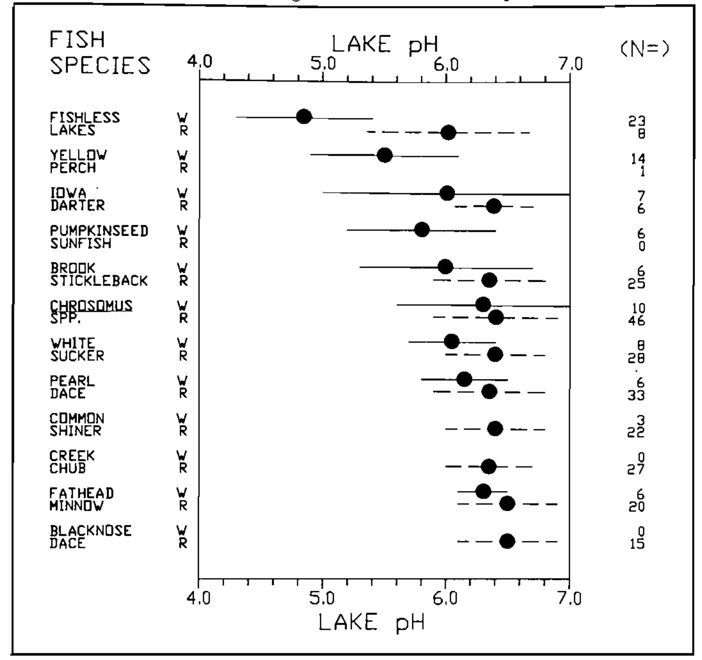


Figure 18
Distribution of individual fish species occurrences (mean ± 1 SD) in relation to lake pH in the Ranger Lake (R) and Wanapitei (W) study areas



declines from 6 to 5, many changes occur in lower trophic levels of the aquatic food web. The results of small non-game fish sampling undertaken in the present study substantiate the acid-tolerance levels noted in other population surveys and/or toxicity tests. For example, the fathead minnow is a relatively intolerant species, often disappearing entirely below pH 6 (Mount 1973; Schindler and Turner 1982; Rahel and Magnuson 1983; Zischke *et al.* 1983), and is extremely vulnerable to the effects of various pollutants (Veith *et al.* 1983; Lemly and Smith 1985). Holtze and Hutchinson (1987) found that the common shiner was the most sensitive of six native fish species tested for aluminum and hydrogen ion stress, whereas Mills (pers. comm.) noted the disappearance of common shiners at an early stage (ca. pH 5.8) in the experimental acidification of Lake 223 at the Experimental Lakes Area. In the same study, pearl dace were capable of surviving below pH 5.6, but suffered adverse effects at pH 5.1 (Mills and Schindler 1986). The yellow perch appears to be tolerant of a wide range of lake pH, although recruitment failures and poor condition have been observed under extremely acidic conditions (Ryan and Harvey 1980; Suns *et al.* 1980; Rask 1984*a*, 1984*b*; Frenette *et al.* 1987).

Notwithstanding the importance of pH as an indicator of the chemical and fishery status of head-water lakes, many other abiotic factors may also influence fish community structure. Fish assemblages, and the physical and chemical factors that influence their organization, have been studied extensively in lakes in northern Wisconsin (Tonn and Magnuson 1982; Rahel 1984, 1986; Tonn 1985). In our study, data on both physical and chemical variables have been analysed for evidence of those factors, including pH, that are responsible for maintaining the structural organization of small non-game fish assemblages. Scores generated in the combined principal factor analysis on all study lakes (Table 7*a*) were used to assess relationships between major fish assemblages and environmental descriptors. Differences between the average scores of cyprinid, non-cyprinid and fishless groups, and individual factors, along with comparisons among fish groups for each factor independently, are presented in Table 13. Factor 1 "acidity" explains the distribution of non-game fish populations with respect to the chemical properties associated with acidification. Cyprinids, in general, are not found in acid conditions; their occurrence along the acidity factor was significantly different from the average scores of either non-cyprinid or fishless groups (Table 13). In Figure 19, the average scores of major fish assemblage types are projected onto the three major environmental factors. At Wanapitei, increasing acidity clearly coincides with the loss of sensitive fish populations (cyprinids) and the increased occurrence of tolerant species (non-cyprinids) and fishless lakes.

Other factors also influence the occurrence and distribution of these species. Whereas factor 2, ionic strength, and factor 3, eutrophication, do not seem to influence major fish assemblage types (Table 13), significant differences were found on factor 4, morphometry. On average, fishless lakes have significantly lower scores on factor 4 than do either cyprinid or non-cyprinid lakes. These results suggest that small lakes, with confined drainage basins, are not only pre-disposed to surface water acidification, but are also more likely to lack fish purely as a result of biogeographic considerations. Large lakes, which have substantial catchment basin areas, are more likely to support diverse cyprinid assemblages.

Lake morphometry also distinguished pond from stream cyprinids at Ranger Lake, with stream species preferring larger, well-drained lakes (Fig. 19). The physical and chemical characteristics responsible for the separation of pond from stream cyprinids at Ranger Lake have been reported elsewhere (Bendell and McNicol 1987*b*). In that analysis, cyprinid species were found to be added to the head-water pond species assemblage as drainage basin size increased (Fig. 20). We concluded that the addition of those species can be accounted for by the increased availability of stream habitat in lakes downstream. Other studies have found significant relationships between the diversity of fish communities and the size of lakes in acidified (Harvey 1975; Somers and Harvey 1984) and unacidified (Harvey 1982; Tonn and Magnuson 1982) areas of eastern North America. Changes in fish assemblage structure at Ranger Lake, related to drainage basin size, coincided with chemical changes that reflect geochemical processes within the watershed and yet were not themselves responsible for limiting the distribution of stream cyprinids.

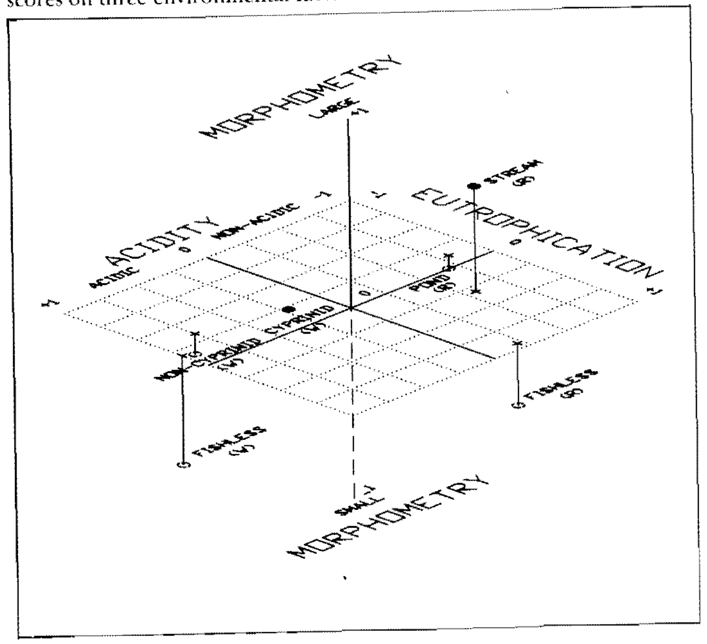
From the evidence of sampling conducted at Ranger Lake in 1981, blacknose dace and fathead minnow populations were believed to be largely restricted to the VIXEN area (see App. 3). This, however, was not found to be true

Table 13
Differences between the average occurrence of major fish assemblage types and individual factor scores in the two study areas combined, tested by Studentized *t*-test procedures (*** $p<0.001$, ** $p<0.01$)†

Fish assemblage	N	Factor 1 "acidity"	Factor 2 "ionic strength"	Factor 3 "eutrophication"	Factor 4 "morphometry"
A Cyprinid	77	− 0.494***	+ 0.055	+ 0.029	+ 0.231
B Non-cyprinid	15	+ 0.760***	+ 0.094	− 0.316	− 0.115
C Fishless	31	+ 0.859***	− 0.182	+ 0.080	− 0.518***
ANOVA		$F = 45.11$ ***	$F = 0.73$	$F = 1.06$	$F = 8.82$ **
Tukey's		A < B,C			A > C

† Comparisons among average factor scores for fish assemblage types were made using Tukey's Studentized range test.

Figure 19
Patterns of association among major fish assemblage types in the Ranger Lake (R) and Wanapitei (W) study areas, as depicted by their average scores on three environmental factors



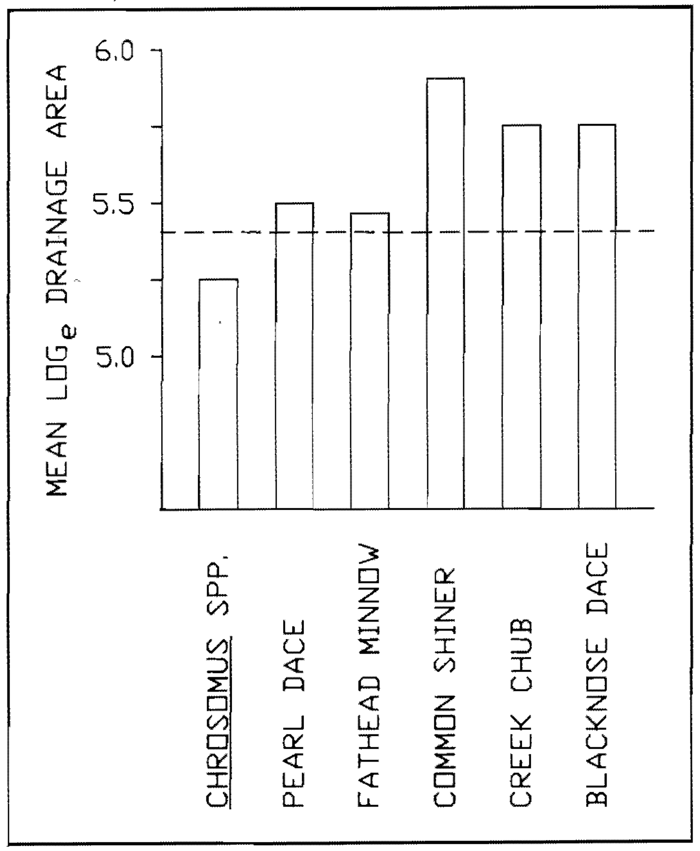
from the broader sampling completed in 1983. There is no reason to believe that blacknose dace and fathead minnow were restricted by pH and alkalinity within the range of values observed at Ranger Lake. Irrespective of geographic location (VIXEN or GONG), lakes with small drainage areas tended to support a more restricted cyprinid fauna and had a lower average buffering capacity, at a given pH, than lakes with larger drainage areas and a more diverse cyprinid fauna (Bendell and McNicol 1987*b*). Those lakes also had higher Ca concentrations, which may mitigate the effects on fish of low pH (Brown 1982).

These results suggest that fish populations in head-water lakes are more likely to suffer losses at a given level of acid deposition than lakes with large basins and similar soils and geology. Under a comparable range of drainage basin characteristics at Wanapitei, fish species richness was highly correlated with lake pH, although both chemical and physical parameters, including drainage basin size, contributed to the gradient of fish community structure observed. Similarly, Rahel (1986) noted that both chemical (alkalinity) and biogeographic (lake size and isolation from other water bodies) factors strongly influenced fish community composition in northern Wisconsin lakes. He concluded that low alkalinity and associated chemical conditions (e.g., low pH) limited the occurrence of many cyprinids and some percids (genus *Etheostoma*), whereas other species were uncommon in small lakes (e.g., smallmouth bass and log perch) or lakes without tributaries (redhorses, genus *Moxostoma*). Other species, including white sucker, sunfish (*Lepomis* spp.) and yellow perch, were present across a wide range of lake size and alkalinity.

3.3. Macroinvertebrates and lake acidification

Acidification of lakes in regions affected by acid deposition can result in substantial changes to communities of planktonic and benthic organisms (Almer *et al.* 1974; Hendrey *et al.* 1976; Kwiatkowski and Roff 1976; Weiderholm and Eriksson 1977; Mossberg and Nyberg 1979). Changes in the occurrence, abundance, seasonal succession of species and diversity of the community have been used as

Figure 20
Mean log of drainage area for each of the major cyprinid taxa in the Ranger Lake study area compared with the overall mean (horizontal dashed line)



indicators of lake acidification. Kelso *et al.* (1986*a*) showed that the trend of fewer species of fish, phytoplankton, zooplankton and benthos below pH 6 persists throughout eastern Canada, although the least response is seen in the benthos where certain organisms receive the benefits of buffering from their substrate (Dermott *et al.* 1986). However, Kelso *et al.* (1986*a*) also concluded that the evidence from eastern Canada indicates that the physical limits of the aquatic habitat, further affected by pH and alkalinity, exert a strong influence upon community diversity of all the biotic components.

Macrofauna of both the benthic and nektonic communities were examined in the 1981 pilot study conducted at Ranger Lake (see App. 3). Aquatic invertebrate sampling conducted in 1983 focussed entirely on epifaunic and nektonic organisms considered to be of potential value as waterfowl foods. Although many benthic invertebrates, such as species of snails, clams, crayfish, amphipods and various aquatic insects, are seldom found in acidic lakes, certain large predaceous aquatic insects, such as water boatmen and backswimmers, are very acid-tolerant (Grahn *et al.* 1974), and may become the top predators in acidified lakes (Eriksson *et al.* 1980; Dillon *et al.* 1984).

The composition of aquatic insect assemblages was examined in relation to fish predation and lake acidity on a subset of 18 lakes in the two study areas. Fish were not recorded in 4 lakes examined at Ranger Lake and 8 lakes at Wanapitei. Among the 24 lakes with fish, a total of 12 fish species was recorded (see App. 8). The number of fish captured ranged from 1 to 3857, with fewer than 100 individuals collected on 10 lakes sampled.

Table 14
Presence of major insect taxa in sweep net samples from 18 lakes in the Ranger Lake study area*

Insect taxa†		Lake No. *014R	044R	*436R	*459R	426R	*013R	416R	003R	004R	051R	409R	456R	404R	007R	453R	438R	033R	010R
Cor	*Dasycorixa hybrida*	+	·	+	·	·	·	·	·	·	·	·	·	·	·	·	·	·	·
Not	*Notonecta insulata*	+	·	+	+	·	+	·	·	·	·	·	·	·	·	·	·	·	·
Dip	*Chaoborus americanus*	+	+	·	+	·	+	·	·	·	·	·	·	·	·	·	·	·	·
Cor	*Hesperocorixa scabricula*	·	·	+	+	+	·	·	·	·	·	·	·	·	·	·	·	·	·
Not	*Notonecta undulata*	+	·	+	+	+	·	·	·	·	+	·	·	·	+	·	·	·	·
Col	*Hapliplus*	+	·	·	·	·	·	·	·	·	·	·	·	·	+	·	·	·	·
Tri	*Polycentropus*	+	·	·	+	·	+	·	·	·	·	·	·	·	·	·	·	·	·
Cor	*Sigara decoratella*	+	·	·	·	·	+	·	·	·	·	·	·	·	·	·	·	·	·
Cor	*Sigara penniensis*	·	·	·	+	+	·	·	·	·	·	·	·	·	·	·	·	·	·
Ger	*Gerris comatus*	·	·	·	+	·	·	·	·	·	+	·	·	·	·	·	·	·	·
Nep	*Ranatra*	+	·	·	·	+	·	+	·	·	·	·	·	·	·	·	·	·	·
Tri	*Agrypnia*	+	·	·	·	+	·	·	·	+	+	·	·	·	·	·	·	·	·
Cor	*Sigara solensis*	·	·	·	·	·	·	+	·	·	+	·	·	·	+	·	·	·	·
Eph	*Caenis*	·	·	·	·	·	·	·	+	·	·	·	·	·	+	·	·	·	·
Cor	*Sigara compressoidea*	·	·	·	·	+	+	·	·	·	+	·	·	·	+	·	·	·	·
Tri	*Oxyethira*	·	·	·	·	+	+	·	·	·	·	·	·	·	·	·	·	·	·
Cor	*Sigara mackinacensis*	·	·	·	·	+	·	·	·	·	+	+	·	+	·	·	·	·	·
Ger	*Trepobates inermis*	·	·	·	·	·	·	·	·	+	·	+	+	·	·	·	·	·	·
Odo	Libellulidae	·	·	·	·	+	+	·	·	·	·	+	+	·	·	·	·	·	·
Dip	Chironomini	+	·	·	+	+	+	+	+	+	+	+	+	+	·	+	·	·	·
Dip	Tanytarsini	·	·	·	·	+	+	+	·	+	·	+	+	·	·	+	·	·	+
Dip	Orthocladiinae	+	·	·	·	+	+	·	+	·	+	+	·	·	·	+	·	·	·
Odo	Coenagrionidae	+	·	·	+	+	+	+	+	+	+	·	+	·	·	·	·	·	+
Dip	Ceratopogoninae	·	·	·	·	+	+	·	·	·	·	+	+	·	·	+	·	·	·
Not	*Buenoa*	+	·	+	+	+	+	·	·	+	+	·	·	·	·	·	·	·	+
Col	*Graphoderus liberus*	+	+	+	+	+	+	·	·	·	·	·	·	·	·	·	·	·	+
Col	*Gyrinus*	+	·	·	+	+	+	+	·	·	+	·	·	·	+	·	·	·	+
Ger	*Rheumatobates rileyi*	+	·	+	+	·	·	·	·	·	+	+	+	·	+	+	+	+	+
Ger	*Metrobates hesperius*	·	·	·	·	·	·	·	·	·	·	·	·	·	·	+	+	+	+
Dip	Pentaneurini	+	·	·	+	+	+	·	·	·	·	+	·	·	·	+	·	·	·
Odo	*Aeshna*	·	·	·	·	·	·	·	·	+	·	·	·	·	+	+	·	·	·

* Arrangement of lakes and taxa by TWINSPAN. An asterisk (*) denotes fishless lakes.

† *Abbreviations:*

Insect Orders	Families of Hemiptera
Eph – Ephemeroptera	Cor – Corixidae
Odo – Odonata	Not – Notonectidae
Col – Coleoptera	Ger – Gerridae
Tri – Trichoptera	Nep – Nepidae
Dip – Diptera	

The results of the macroinvertebrate studies are reported elsewhere (Bendell 1986; Bendell and McNicol 1987*a*) and summarized in this paper. Matrices of presence–absence scores for the major aquatic insect taxa for each area were analysed using the TWINSPAN program (Hill 1979*a*). Ordered two-way tables (lakes by insect taxa) were produced for both the Ranger Lake (Table 14) and Wanapitei (Table 15) study areas. Insect assemblages from fishless lakes in both areas, along with those lakes containing only a few fish, shared many more taxa in common than those from lakes containing large numbers of fish. These taxa were primarily nektonic, and included backswimmers (Notonectidae), water boatmen (Corixidae), the dytiscid beetle *Graphoderus liberus*, and the dipteran *Chaoborus americanus*.

Using the DECORANA program (Hill 1979*b*), a detrended correspondence analysis was performed to obtain ordination scores for each lake. A description and discussion of the use of this ordination technique are given by Hill and Gauch (1980). Separate ordinations of the 18 lakes at Ranger Lake and at Wanapitei contrasted fishless or nearly fishless lakes with those containing fish. When lake pH is plotted against first axis ordination scores, it is evident that a strong positive relationship existed at Wanapitei ($r_s = 0.79, p < 0.001$) (Fig. 22) but not at Ranger Lake ($r_s = -0.006, p > 0.05$) (Fig. 21). An ordination combining

Table 15
Presence of major insect taxa in sweep net samples from 18 lakes in the Wanapitei study area*

Insect taxa†		Lake No. 002W	*258W	003W	*401W	*572W	*902W	*013W	*251W	*316W	*402W	256W	583W	240W	404W	197W	333W	409W	905W
Meg	*Chauliodes*	·	·	·	·	·	·	+	·	+	+	·	·	·	·	·	·	·	·
Cor	*Sigara decoratella*	·	·	+	+	+	·	·	·	·	·	·	·	·	·	·	·	·	·
Col	*Dineutes nigrior*	·	·	+	+	+	+	·	·	·	·	·	·	·	·	·	·	·	·
Cor	*Sigara dolabra*	·	·	·	·	+	·	·	·	+	·	·	·	·	·	·	·	·	·
Cor	*Sigara penniensis*	·	·	·	+	+	·	·	·	+	·	·	·	·	·	·	·	·	·
Cor	*Hesperocorixa scabricula*	·	·	+	·	·	+	·	·	+	·	·	·	·	·	·	·	·	·
Not	*Notonecta insulata*	·	·	·	+	·	+	·	·	+	+	·	·	·	·	·	·	·	·
Tri	*Triaenodes*	·	·	·	+	·	+	·	·	·	·	·	·	·	·	·	·	·	·
Col	*Graphoderus liberus*	·	+	+	+	·	+	+	·	+	·	·	·	·	·	·	·	·	·
Col	*Coptotomus*	·	+	·	·	·	·	+	·	·	·	·	·	·	·	·	·	·	·
Dip	*Chaoborus americanus*	·	+	+	·	·	·	+	·	+	+	+	·	·	·	·	·	·	·
Odo	Libellulidae	·	+	·	·	+	·	·	·	+	+	·	·	·	·	·	·	·	·
Not	*Notonecta undulata*	·	+	+	+	+	+	+	+	+	·	·	·	·	·	·	·	·	·
Not	*Buenoa*	·	+	+	+	+	+	+	+	+	+	·	·	·	·	·	·	·	·
Dip	Orthocladiinae	+	·	·	·	+	+	+	+	+	+	+	·	·	·	·	·	·	·
Odo	Coenagrionidae	+	·	+	·	+	+	+	·	+	+	·	·	·	·	·	·	·	·
Col	*Gyrinus*	·	+	+	+	+	+	+	·	+	+	+	·	+	+	·	·	·	·
Dip	Pentaneurini	·	+	+	·	+	·	+	·	+	·	+	+	+	·	·	·	·	·
Dip	Tanytarsini	·	·	+	·	+	+	+	+	+	+	+	·	+	·	+	·	·	·
Tri	*Oxyethira*	·	·	·	·	·	·	+	·	+	+	+	+	+	·	·	·	·	·
Dip	Ceratopogoninae	·	·	·	·	+	·	·	·	+	+	+	·	·	+	+	·	·	·
Dip	Chironomini	·	·	+	·	+	+	+	+	+	+	·	·	+	+	+	·	·	+
Ger	*Rheumatobates rileyi*	·	·	·	·	·	·	·	·	+	+	+	·	+	+	+	+	+	+
Ger	*Metrobates hesperius*	·	·	·	·	·	·	·	·	·	·	+	·	·	+	·	+	+	·
Eph	Baetidae	+	·	·	·	·	·	·	·	·	·	·	·	·	·	+	·	+	·
Ger	*Gerris comatus*	·	+	·	·	·	·	·	·	·	·	·	·	+	+	·	+	·	·
Tri	*Polycentropus*	·	+	·	·	·	·	·	·	·	·	·	·	·	+	·	·	·	·

* Arrangement of lakes and taxa by TWINSPAN. An asterisk (*) denotes fishless lakes.

† *Abbreviations:*

Insect Orders
Eph – Ephemeroptera
Odo – Odonata
Meg – Megaloptera
Col – Coleoptera
Tri – Trichoptera
Dip – Diptera

Families of Hemiptera
Cor – Corixidae
Not – Notonectidae
Ger – Gerridae

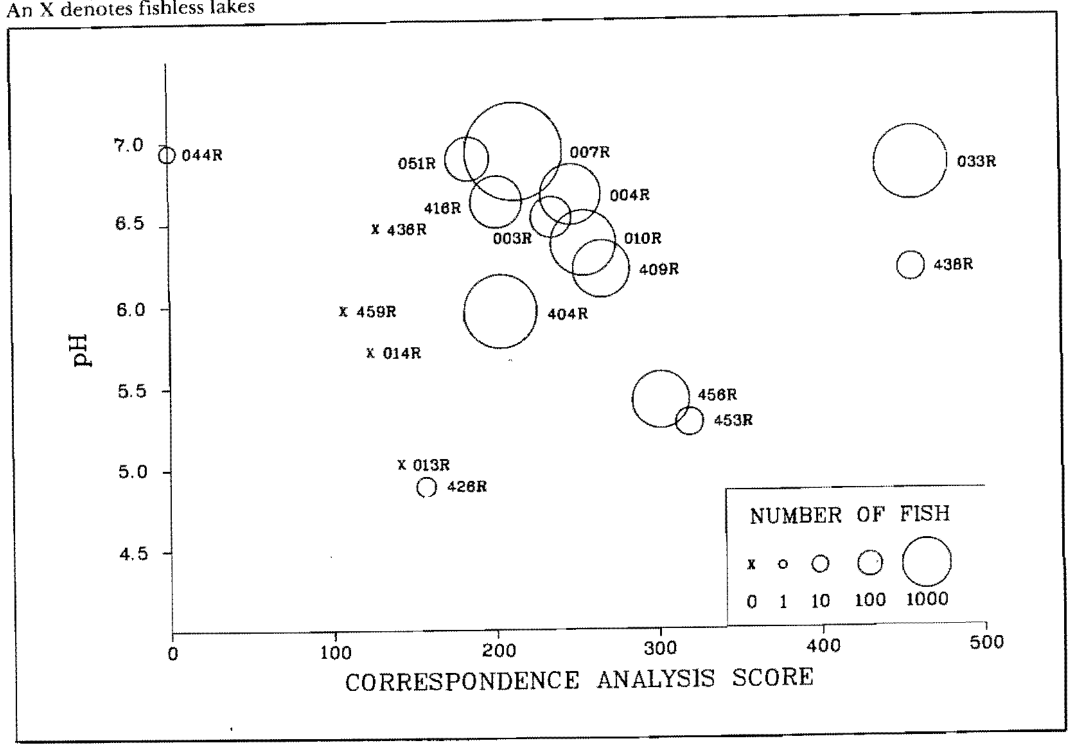
lakes from both areas showed that fishless lakes had similar aquatic insect assemblages (Fig. 23), although the range in pH of fishless lakes at Ranger Lake (6.45–5.0) was higher than that observed at Wanapitei (5.27–4.44). The ordinations indicate, therefore, that fishless or nearly fishless lakes support a similar aquatic insect assemblage distinct from lakes with fish, and that the fishless lake assemblage occurs irrespective of lake pH. At Wanapitei, differences in aquatic insect assemblages were likely related to lake pH as a probable consequence of the effects of lake acidification on fish populations.

Aquatic insect species richness was also significantly greater in fishless lakes (Mann-Whitney U tests, $p < 0.05$). The average number of major taxa identified was more than twice as high in fishless lakes at Wanapitei and Ranger Lake (11.9 and 13.5, respectively) than in lakes containing fish (5.3 and 6.0, respectively).

To test the hypothesis that more aquatic insects were available to sweep net sampling in lakes without fish than in lakes with fish, contrasts of means of log transformed numbers in net samples from the two lake types were performed. Highly significant differences were found at Ranger Lake (t = 7.39, $p < 0.001$) and at Wanapitei (t = 5.96, $p < 0.001$). The greater number of insects in samples from fishless lakes was largely the result of the abundance of nektonic organisms (Fig. 24). However, several lakes with fish and without nekton nonetheless recorded numbers of insects comparable with those in fishless lakes because of large numbers of waterstriders (Gerridae), particularly *Rheumatobates rileyi*, and less often *Metrobates hesperius*.

Lakes with and without fish in the Wanapitei area generally supported two distinct and very separate types of insect assemblages (Figs. 22 and 24), whereas lakes at Ranger Lake tended to have a broader degree of overlap between the same extremes (Figs. 21 and 24). Two major factors

Figure 21
Lake pH versus first axis correspondence analysis scores (DECORANA) calculated on presence-absence data for 31 insect taxa in lakes with and without fish in the Ranger Lake (R) study area. The size of the open circles is proportional to the number of fish taken in five minnow traps. An X denotes fishless lakes



accounted for such differences. First, at Wanapitei, the waterstrider *R. rileyi* was often abundant on lakes with fish and scarce on acidic fishless lakes, whereas at Ranger Lake it was recorded in large numbers on three of four non-acidic (pH > 5.5) fishless lakes. Its absence from acidic lakes suggests that it is affected by acidification independently of the effects of fish predation. Secondly, as outlined earlier, major differences in fish faunas occur between the two areas. Half of the lakes containing fish at Wanapitei support yellow perch, which is a particularly effective insectivore (Eriksson 1979; Post and Cucin 1984), and which is also acid-tolerant. At Ranger Lake, the fauna of small non-game fish is characterized by an assemblage of cyprinids, with yellow perch recorded in only one lake. Cyprinids are less likely to be effective predators of insects because several of them, including *Chrosomus* spp. and fathead minnow, are gape-limited predators of the large aquatic insects and must, therefore, prey only on their early instars. Also, this assemblage is characteristic of lakes subject to winter kill (Tonn and Magnuson 1982). One can expect fish populations in such lakes to be limited by winter oxygen levels or the availability of stream refugia from anoxic conditions. Such conditions occurred more often at Ranger Lake, where shallow head-waters (ZMAX < 3 m) are common and therefore likely to develop anoxic conditions. Of those 18 lakes sampled at Wanapitei, there were no shallow head-waters which contained fish, and the average maximum depth (7.7 m) was twice that at Ranger Lake (3.8 m).

Figure 22
Lake pH versus first axis correspondence analysis scores (DECORANA) calculated on presence-absence data for 27 insect taxa in lakes with and without fish in the Wanapitei (W) study area. The size of the open circles is proportional to the number of fish in five minnow traps. An X denotes fishless lakes

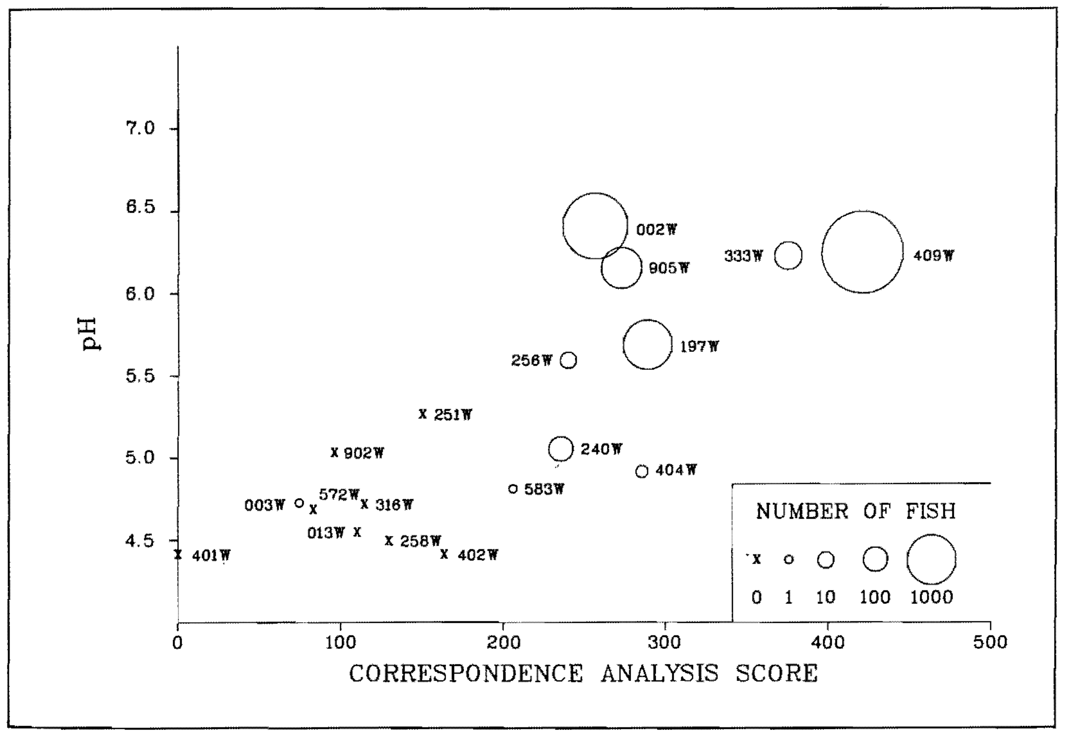


Figure 23
First axis correspondence analysis scores (DECORANA) calculated on presence-absence data for 37 insect taxa in lakes in the Ranger Lake (R) and Wanapitei (W) study areas. Open and closed circles represent lakes with and without fish, respectively

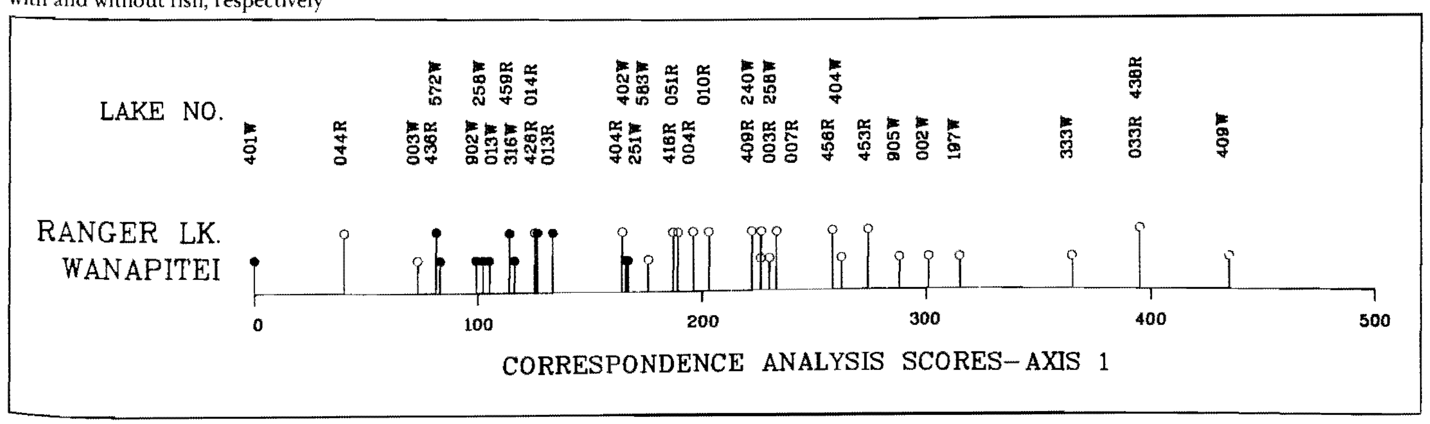
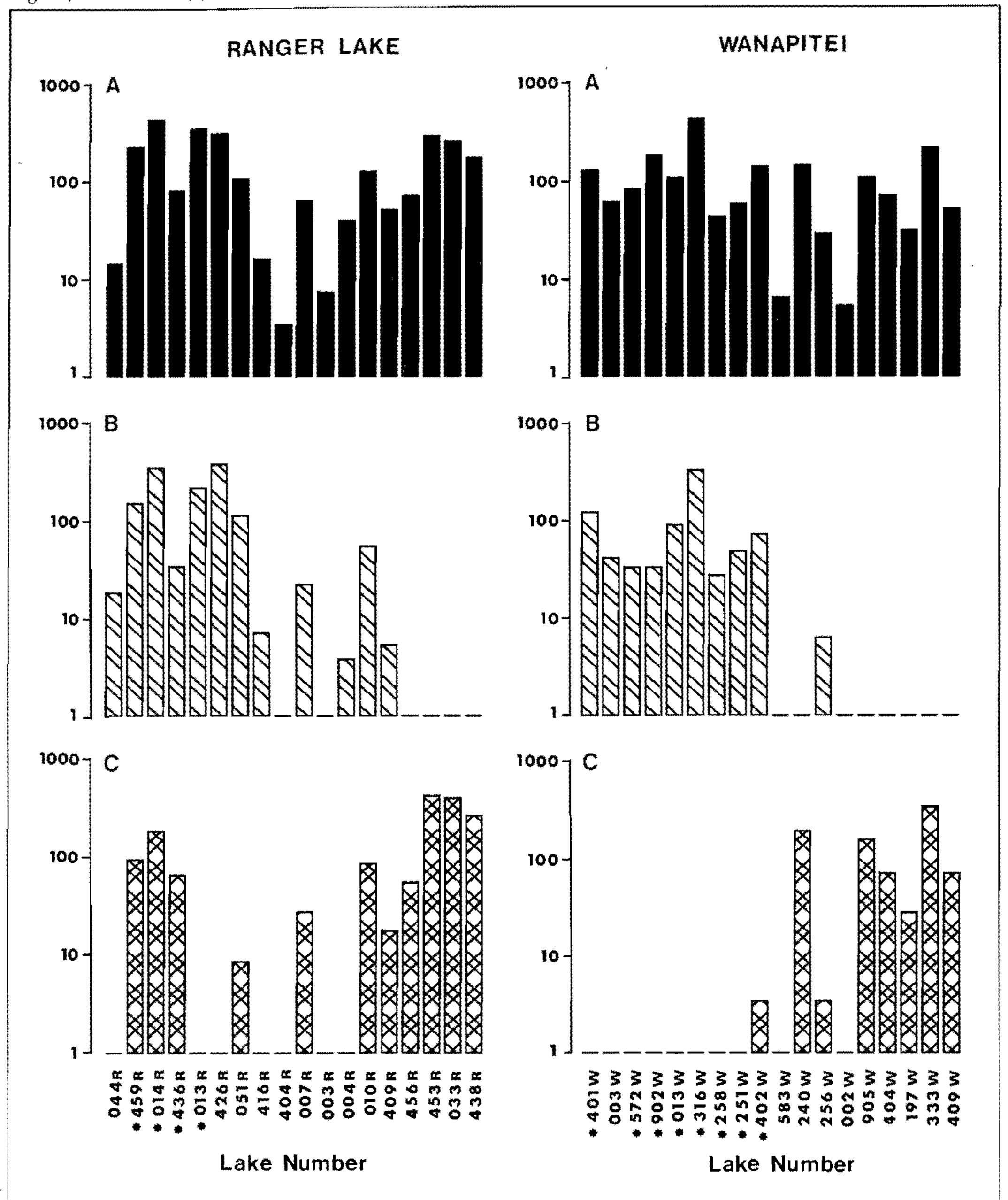


Figure 24
Log abundance of (*a*) all aquatic insects, (*b*) nekton (sum of the numbers of corixids, notonectids, *G. liberus* and *C. americanus*), and (*c*) Gerridae in the Ranger Lake (R) and Wanapitei (W) study areas. Lakes are ordered by increasing first axis correspondence analysis scores (DECORANA) (see Fig. 23). An asterisk (*) denotes fishless lakes



The results of this study support the postulations of Eriksson *et al.* (1980) that many of the changes in the macroinvertebrate assemblages that coincide with acidification are related to changes in predator–prey relations and are not direct toxic effects of lowered pH on the macroinvertebrates themselves. In general, fish select the largest and most conspicuous food organisms in the water column (Brooks 1968; Werner and Hall 1974; Zaret 1980). Often fish eliminate large crustacean zooplankters (Brooks and Dodson 1965; Galbraith 1967; Wells 1970), as well as many large, active nektonic insects, such as some *Chaoborus* spp. (Pope *et al.* 1973; Northcote *et al.* 1978; von Ende 1979; Mossberg and Nyberg 1979), Corixidae and Notonectidae (Macan 1965; Weir 1972; Henrikson and Oscarson 1978; Hurlbert and Mulla 1981). Fish can also have a great effect on other components of the fresh-water community (Thorp 1986). Significant relationships have been identified between fish and phytoplankton (Hrbacek *et al.* 1961; Lynch and Shapiro 1981), benthos (Hall *et al.* 1970; Gilinsky 1984; Post and Cucin 1984), macrophytes (Spencer and King 1984) and waterfowl usage (Eriksson 1979, 1985, 1986; Eadie and Keast 1982). In fishless lakes, predatory insects become the top predators in the limnetic zone. This insect assemblage is a naturally occurring ensemble of species that can be found in small lakes and ponds where biogeographic isolation, winter anoxia, overfishing or natural (organic) acidity excludes fish. Given that a limited number of aquatic insect taxa are as acid-sensitive as fish, changes in the abundance of particular species, or changes in overall community structure, must be examined with respect to changes in fish populations. Where acidification has occurred, similar structural changes will occur at different levels of pH depending on the

sensitivity of the particular fish species involved. In the Wanapitei area, acid lakes containing populations of acid-tolerant yellow perch (404W, 583W and 240W) did not support populations of nektonic insects that were abundant in fishless lakes at similar pH levels.

3.4. Lake characteristics and waterfowl habitat selection

Data from aerial and ground surveys were used to examine the differences in breeding success and habitat utilization by adult and young waterfowl. The frequency of occurrence of waterfowl during the nest-initiation period, as well as sightings of broods and adults during the brood-rearing period, are summarized in Table 16. A comparison of the brood and indicated nesting pair occupancy rates is presented in Table 17. Indicated nesting pair estimates for Ranger Lake were derived from surveys conducted in 1981 only, whereas estimates at Wanapitei were taken from 1983 surveys (see Results and Discussion 1.2. for discussion of temporal comparability of indicated nesting pair results).

Seven species were commonly recorded in aerial and ground surveys. More than two-thirds (67%) of the 95 lakes surveyed during the nest-initiation period in both study areas combined contained at least one indicated nesting pair (Ranger Lake, 59%; Wanapitei, 76%) (Table 16). These occupancy levels are substantially higher than those observed in systematic aerial surveys conducted in corresponding blocks (Sault Ste. Marie, 34%; New Liskeard, 45%) (Table 3). However, fewer than one-third (31%) of the lakes surveyed actually supported broods (Table 16). The Common Loon was the most frequently encountered species during both the nest-initiation and brood-rearing periods, followed by Ring-necked Duck and Common Goldeneye, with Hooded Merganser, American Black Duck and Common Merganser also being seen regularly. Mallards were observed infrequently.

Few significant differences in occupancy levels were observed between the two study areas (Table 16). On average, slightly more study lakes were occupied by waterfowl at Wanapitei than at Ranger Lake during the nest-initiation period (χ^2 = 3.18, $p < 0.10$). The opposite trend was observed during the brood-rearing period, with slightly fewer lakes containing broods at Wanapitei than at Ranger Lake (χ^2 = 3.67, $p < 0.10$).

Table 16
Differences in the number and percentage of lakes occupied by seven waterfowl species and combined taxonomic groups in the Ranger Lake and Wanapitei study areas, expressed as indicated nesting pairs, combined adult and/or brood sightings during the brood rearing period, and brood sightings only, tested using chi-squared tests of independence or Fisher exact probability test (**$p<0.05$, *$p<0.10$)

	Indicated pairs				Adults and broods				Broods only			
	Ranger Lake (N=49)		Wanapitei (N=46)		Ranger Lake (N=68)		Wanapitei (N=54)		Ranger Lake (N=68)		Wanapitei (N=54)	
Species	No.	%	No.	%	No.	%	No.	%	No.	%	No.	%
1. Common Loon	12	25	15	33	24	35	22	41	4	6	1	2
2. Common Merganser	4	8	3	7	13	19	5	9	5	7	0	0*
3. Common Goldeneye	8	16	10	22	16	24	13	24	9	13	5	9
4. Hooded Merganser	4	8	6	13	15	22	10	19	5	7	5	9
5. Ring-necked Duck	6	12	14	30**	20	29	10	19	10	15	3	6
6. American Black Duck	8	16	8	17	6	9	4	7	1	2	3	6
7. Mallard	4	8	3	7	3	4	0	0	1	2	0	0
Piscivores (1, 2)	14	29	16	35	33	48	23	43	9	13	1	2**
Insectivores (3, 4)	10	20	15	33	24	35	20	37	14	20	8	15
Omnivores (5, 6, 7)	14	29	20	44	22	32	12	22	11	16	5	9
Total duck (2–7)	21	43	28	61*	45	66	28	52	24	35	11	20*
Total waterbird (1–7)	29	59	35	76*	52	77	40	74	26	38	12	22*

Table 17
The number and percentage of lakes occupied by indicated nesting pairs (IP) and broods (BR) of seven waterfowl species and combined taxonomic groups in the Ranger Lake and Wanapitei study areas, as well as ratios of brood versus indicated nesting pair occupancy

	Ranger Lake (N=49)				Wanapitei (N=46)				Brood: indicated pair (BR:IP ratio)	
	IP		BR		IP		BR			
Species	No.	%	No.	%	No.	%	No.	%	Ranger Lake	Wanapitei
1. Common Loon	12	24.5	4	8.2	15	32.6	1	2.2	0.33	0.07
2. Common Merganser	4	8.2	3	6.1	3	6.5	0	–	0.74	0.00
3. Common Goldeneye	8	16.3	7	14.3	10	21.7	5	10.9	0.88	0.50
4. Hooded Merganser	4	8.2	4	8.2	6	13.0	4	8.7	1.00	0.67
5. Ring-necked Duck	6	12.2	7	14.3	14	30.4	3	6.5	1.17	0.21
6. American Black Duck	8	16.3	0	–	8	17.4	2	4.3	0.00	0.25
7. Mallard	4	8.2	1	2.0	3	6.5	0	-	0.24	0.00
Piscivores (1, 2)	14	28.6	7	14.3	16	34.8	1	2.2	0.50	0.06
Insectivores (3, 4)	10	20.4	11	33.0	15	32.6	7	15.2	1.09	0.47
Omnivores (5, 6, 7)	14	28.6	8	44.0	20	43.5	4	8.7	0.57	0.20
Total duck (2–7)	21	42.9	18	36.7	28	60.9	9	19.6	0.86	0.32
Total waterbird (1–7)	29	59.2	20	40.8	35	76.1	10	21.7	0.69	0.29

Table 18
Differences between the average occurrence of (i) indicated nesting pairs during the nest-initiation period, and (ii) combined adult and/or brood sightings during the brood-rearing period, and individual factor scores for seven waterfowl species and combined taxonomic groups in the two study areas combined, tested by Studentized *t*-test procedures
($^{****}p<0.001$, $^{***}p<0.01$, $^{**}p<0.05$, $^{*}p<0.10$)

Species		Factor 1 "acidity"	Factor 2 "ionic strength"	Factor 3 "eutrophication"	Factor 4 "morphometry"
(i) Nest-initiation period	(*N* = 94)				
1. Common Loon	27	+ .148	− .126	− .466***	+ .622****
2. Common Merganser	7	− .186	− .035	− .388	+ .342
3. Common Goldeneye	18	+ .392	− .168	− .110	− .236
4. Hooded Merganser	10	+ .467	− .181	+ .159	+ .107
5. Ring-necked Duck	20	+ .641	− .376	+ .162	− .086
6. American Black Duck	16	+ .165	− .115	+ .239	+ .159
7. Mallard	7	+ .034	− .057	+ .442	+ .105
Piscivores (1,2)	30	+ .158	− .132	− .423***	+ .519***
Insectivores (3,4)	25	+ .415	− .163	− .057	− .170
Omnivores (5,6,7)	34	+ .384*	− .199	+ .149	+ .021
(ii) Brood-rearing period	(*N* = 122)				
1. Common Loon	46	− .089	+ .044	− .418****	+ .448****
2. Common Merganser	18	− .159	+ .256	+ .014	+ .722****
3. Common Goldeneye	29	+ .285*	− .086	+ .292**	− .252*
4. Hooded Merganser	25	− .257	− .069	− .002	− .166
5. Ring-necked Duck	30	− .119	+ .154	+ .423****	+ .052
6. American Black Duck	10	− .129	+ .226	+ .633****	+ .216
7. Mallard	3†				
Piscivores (1,2)	56	− .173*	+ .103	− .263***	+ .483****
Insectivores (3, 4)	44	+ .066	+ .041	+ .059	− .213**
Omnivores (5,6,7)	34	− .081	+ .011	+ .453****	+ .061

† Insufficient sample size.

Many factors, including both chemical and physical parameters, influence the suitability of a lake for breeding waterfowl. Scores generated in the factor analysis were used to assess the relationship between waterfowl and environmental descriptors. Differences between the occurrence of indicated nesting pairs and combined adult and/or brood sightings during the brood-rearing period and individual factors are presented in Table 18. No significant separation was detected amongst any waterfowl group and factor 2, ionic strength.

Few significant differences were found in a comparison of indicated nesting pair estimates. During the nest-initiation period, Common Loons have significantly higher scores on factor 4, morphometry, and lower scores on factor 3, eutrophication. The preference shown by Common Loons for larger, more oligotrophic lakes is consistent with the results of broader systematic aerial surveys conducted in north-eastern Ontario (Fig. 8).

With the exception of Common Mergansers, all species had moderately high positive loadings on the acidity factor during the nest-initiation period (Fig. 25). The pronounced acidity of Wanapitei lakes, coupled with the higher overall occupancy levels observed at Wanapitei (76%) compared with Ranger Lake (59%), partially explains this trend. The results suggest that the distribution of indicated nesting pairs during spring aerial surveys was positively correlated with the acidity of the lakes on which they were recorded. During the brood-rearing period, significant associations between waterfowl and abiotic environmental descriptors were also found. With the exception of Common Goldeneye and Common Merganser, the average score of lakes occupied by other species during the brood-rearing period shifted towards the non-acidic portion of the gradient (Fig. 25), suggesting that lakes occupied by broods tend to be less acidic than those used during the nest-initiation period. These differences were most pronounced for the Ring-necked Duck and Hooded Merganser: average scores for lakes where only indicated nesting pairs were seen were significantly more acidic than those lakes where adults and/or broods were seen during the brood-rearing period (ANOVA: $F = 5.40$, $p < 0.05$, and $F = 6.22$, $p < 0.05$, respectively).

Biotic factors, especially the occurrence of fish, also influence the distribution and ultimate success of waterfowl broods and must be considered in any analysis of waterfowl habitat selection. The results of brood surveys substantiated the importance of fish in head-water lakes for their ultimate use by certain waterfowl species (Fig. 26), as suspected in the 1981 survey data (see App. 3). Based on the percentage of lakes occupied by adults and/or broods, under both fish and fishless conditions, piscivorous species, including Common Merganser and, in particular, Common Loon, preferred lakes with fish. Conversely, the insectivorous species, Common Goldeneye and Hooded Merganser, showed a distinct preference for fishless lakes, whereas more generalized feeders, including Ring-necked Duck and American Black Duck, showed little preference for either condition.

Patterns of association of waterfowl species, based on adult and/or brood sightings during the brood-rearing period, and major fish assemblage types (cyprinid, non-cyprinid and fishless lakes) are depicted in Figure 27, where the average score for any biological group is projected on the three major environmental factors. To interpret the significance of the habitat and biotic associations depicted, a variable selection process was followed (B. Collins, pers. comm.), where the probability (p_i) that lake i ($i = 1..N$) has a bird present was assumed to be a linear function of the independent variables (factor scores and presence of fish).

Figure 25
Comparison between average scores on the three environmental factors for indicated nesting pairs and adult and/or brood sightings during the brood-rearing period for six waterfowl species in the two study areas combined. The direction of change between average factor loadings for indicated nesting pairs (May surveys) and adult and/or brood sightings (June–July surveys) is indicated by an arrow

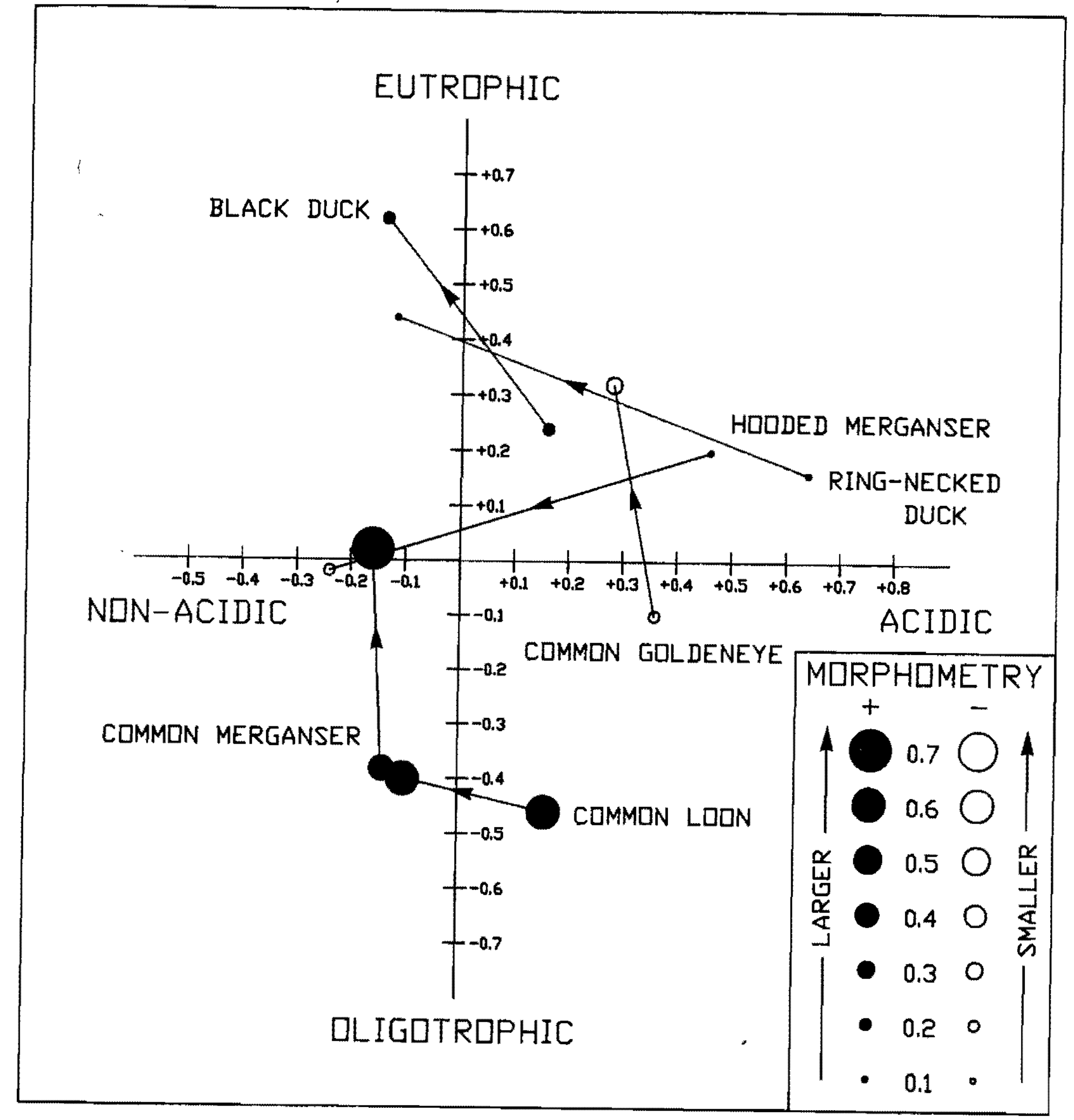
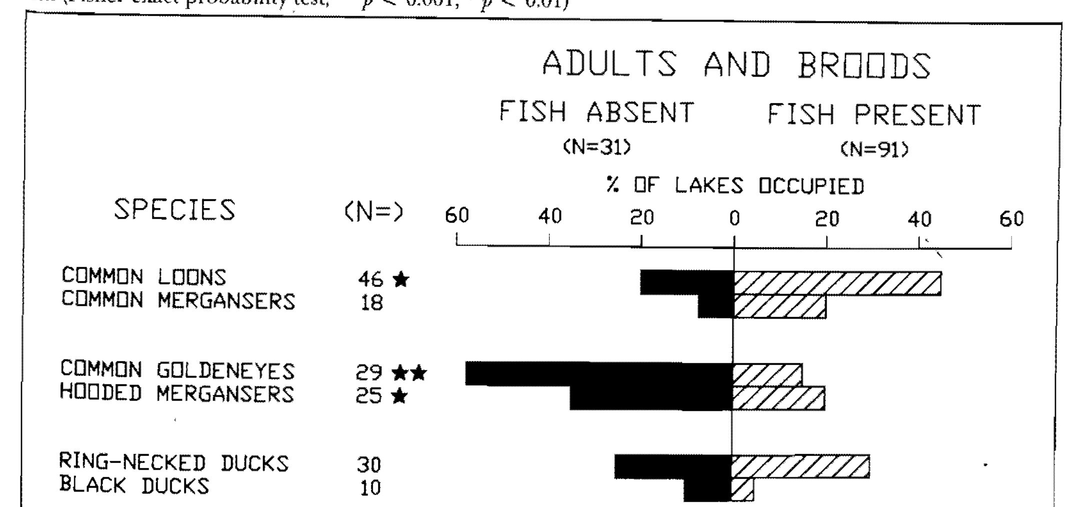


Figure 26
Comparison of adult and/or brood sightings during the brood-rearing period in the two study areas combined, in relation to the occurrence of fish (Fisher exact probability test, ** $p < 0.001$, * $p < 0.01$)



The distribution of the variable y_i, which is 1 if a bird is present on lake i and 0 otherwise, for all N values gives the likelihood equation:

$$L = \sum_{i=1}^{N} p_i^{y_i} (1-p_i)^{(1-y_i)} \quad [4]$$

p_i can be modelled as a linear function:

$$p_i = b_o + b_i x_i \quad [5]$$

where b_o and b_i are selected to maximize the likelihood function (L). The significance of each variable can be assessed using the likelihood ratio or G test (Sokal and Rohlf 1981). The analysis was performed in a stepwise manner separately for each waterfowl species, with equation [4] fitted five times with x_i set to one of the four factor scores or the presence of fish variable. If the G test indicated a significant improvement in the model fitting, then the variable which caused the greatest improvement was selected, and the curve fitting proceeded. Otherwise, the variable selection process was terminated. In the next step, the following model was fitted:

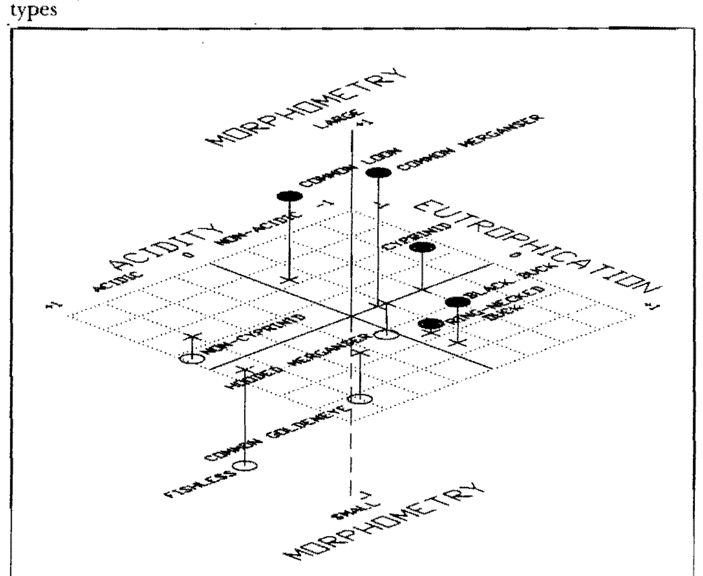
$$p_i = b_o + b_1 * x_i + b_2 * y_i \quad [6]$$

where x_i denotes the variable selected in equation [4] and y_i one of the other variables. This model was fitted for each of the four variables not selected in the first step. However, if fish presence was one of the variables in equation [5], then an interaction term, which indicates whether the effect of the factor variable is the same for lakes with and without fish, was included in the model:

$$p_i = b_o + b_1 * x_i + b_2 * y_i + b_3 * x_i * y_i \quad [7]$$

If the G test indicated that the likelihood equation increased significantly by including a second variable, then a model involving three terms was fitted. The stepwise selection was continued until no further significant increases in the likelihood equation were made.

Figure 27
Patterns of association in the two study areas combined among average scores on the three environmental factors for adult and/or brood sightings during the brood-rearing period and major fish assemblage types



The results of this stepwise analysis are summarized in Table 19, with each species represented by the best-fitting linear model for the likelihood equation. The presence of small non-game fish was not, in itself, the major factor influencing the distribution of piscivorous species, either Common Loon or Common Merganser, in the lakes studied. Fish presence was among three variables selected in step 1 of the Common Loon analysis, but was found to have a significant ($p < 0.05$) interaction with factor 3 in the second step. Both Common Loon and Common Merganser occurred most frequently on lakes with high factor 4 scores, suggesting that they preferred large lakes with irregular shorelines often draining a substantial area. Also, the occurrence of Common Loons was negatively correlated with factor 3, eutrophication, suggesting a preference for the deeper, more dilute lakes, often characterized by low nutrient and organic carbon levels. Also unaffected by fish occurrence were Ring-necked Duck and American Black Duck, both of which preferred the shallow, nutrient-rich wetlands, which were often dominated by riparian and emergent vegetation.

The presence of fish did reduce the likelihood of any lake supporting either Common Goldeneye or Hooded Merganser (Fig. 26), suggesting that fish may act as competitors for insect prey exploited by these ducks. Common Goldeneye occurrence was also positively correlated with factor 3, eutrophication. In much the same way, Hooded Merganser occurrence was negatively correlated with factor 1, acidity, in the second step of the analysis. This suggests that, in addition to preferring fishless conditions, this species also tended to avoid acidic conditions and may favour those non-acidic, naturally fishless lakes (always head-water) which composed a small proportion (<15%) of all habitat studied.

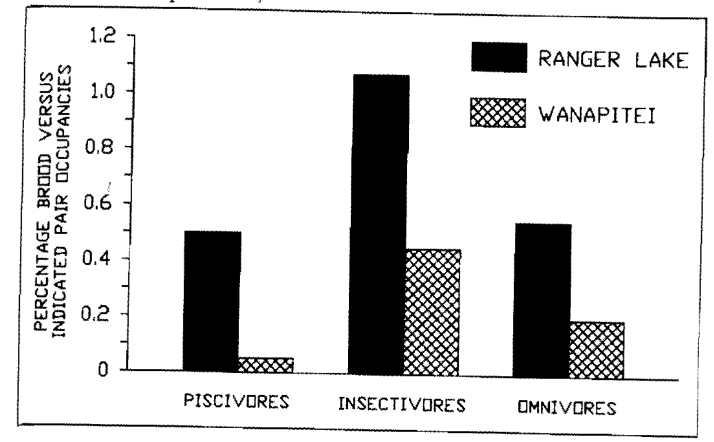
Table 19
Relationships between the distribution of individual waterfowl species, environmental factors and fish occurrence in the two study areas combined as examined using a stepwise variable selection process*

(i) Species	Equation ($p < 0.05$)	
Common Loon	p_i = 0.386 – 0.106 Factor 3 + 0.067 Factor 4	
	(± 0.051) (± 0.037) (± 0.060)	
Common Merganser	p_i = 0.115 + 0.034 Factor 4	
	(± 0.025) (± 0.016)	
Common Goldeneye	p_i = 0.563 + 0.050 Factor 3	(if fish not present)
	(± 0.105) (± 0.012)	
	p_i = 0.125 + 0.050 Factor 3	(if fish present)
	(± 0.039) (± 0.012)	
Hooded Merganser	p_i = 0.386 – 0.078 Factor 1	(if fish not present)
	(± 0.036) (± 0.030)	
	pi = 0.130 – 0.078 Factor 1	(if fish present)
	(± 0.036) (± 0.030)	
Ring-necked Duck	p_i = 0.250 – 0.100 Factor 3	
	(± 0.047) (± 0.014)	
American Black Duck	p_i = 0.090 – 0.036 Factor 3	
	(± 0.023) (± 6.0 E-8)	

(ii) Species	N	Factor 1 "acidity"	Factor 2 "ionic strength"	Factor 3 "eutrophication"	Factor 4 "morphometry"	Fish present
Common Loon	46			– (1)†	+ (2)	
Common Merganser	16				+ (1)	
Common Goldeneye	29			+ (2)		– (1)
Hooded Merganser	25	– (2)				– (1)
Ring-necked Duck	30			+ (1)		
American Black Duck	10			+ (1)		

* Waterfowl data are expressed as combined adult and/or brood sightings during the brood-rearing period. Equations providing the best fit to the linear model tested for each species are presented in part (i). The figures under the parameters are estimates of the standard errors of the parameters. Those significant relationships found are illustrated schematically in part (ii).
† Values in parentheses indicate order of inclusion in the model.

Figure 28
Brood to indicated nesting pair occupancy ratios for three major waterfowl groups (piscivores—Common Loon + Common Merganser; insectivores—Common Goldeneye + Hooded Merganser; and omnivores—Ring-necked Duck + Black Duck + Mallard) in the Ranger Lake and Wanapitei study areas



Despite the pronounced differences in chemical conditions and trophic structure between the two study areas, no significant differences were detected between the two areas in the distribution of nesting waterfowl observed in indicated nesting pair surveys. However, the percentage of lakes occupied by broods, in relation to the percentage of lakes occupied by indicated nesting pairs, does suggest differences in reproductive output per indicated pair between two study areas (Table 17). At Ranger Lake, there was a very good relationship between brood and indicated nesting pair occupancy rates for Common Merganser, Common Goldeneye, Hooded Merganser and Ring-necked Duck, with the brood versus indicated pair (BR:IP) ratio ranging from 0.74 to 1.17. The ratio was substantially lower for Common Loon (0.33), Black Duck (0) and Mallard (0.24). Overall, the ratio between brood and indicated nesting pair occupancy rates was 0.50 for piscivores (Common Loon and Common Merganser), 0.57 for omnivores (Ring-necked Duck, Black Duck and Mallard) and 1.09 for insectivores (Common Goldeneye and Hooded Merganser) (Fig. 28). At Wanapitei, this ratio was generally lower for all species, except Black Duck (Table 17). Both omnivore (0.20) and insectivore (0.47) ratios were substantially lower at Wanapitei than at Ranger Lake, although the most pronounced difference in success rates was shown by piscivores (0.06).

Conclusions

1. Waterfowl populations in north-eastern Ontario

The loss and degradation of habitat are the major waterfowl management problems currently facing management agencies in North America (North American Waterfowl Management Plan 1986). Wetlands in the boreal forest, although supporting lower densities of breeding ducks than those of the Prairie Provinces (Wellein and Lumsden 1964), have become increasingly important to continental waterfowl populations, particularly during droughts and as prairie wetlands are destroyed by agricultural drainage (Gilmer *et al.* 1975). However, these formerly secure waterfowl habitats in the vast boreal forests of eastern Canada are now affected by hydropower and recreational developments, forestry practices, industrial effluent pollution and atmospheric contamination. In eastern North America, the marked decline in Black Duck numbers and suspected drop in the Common Goldeneye and Green-winged Teal populations may be related to boreal habitat degradation (North American Waterfowl Management Plan 1986). On the Precambrian Shield of north-eastern and south-central Ontario substantial input of atmospheric pollutants to sensitive terrain might threaten the quality of aquatic systems for waterfowl production.

Prior to the current work, only one reconnaissance survey (Dennis 1974*b*) had examined breeding distributions of waterfowl in north-eastern Ontario. Consequently, historical data on waterfowl population levels in acid-affected areas are not available. Surveys conducted in the region since 1980 have shown no significant trends in population levels, with the exception of Wood Ducks, which showed a consistent rise in numbers. Such short-term surveys, however, could not possibly establish relationships between lake acidification and waterfowl population responses. Breeding pair surveys have shown that only a portion (<45%) of the aquatic habitat available to nesting waterfowl in the region is being utilized. Several factors, including overhunting and habitat degradation, could account for this situation; certainly no one factor alone is responsible for the rather low population densities (about one pair per square kilometre) observed throughout north-eastern Ontario. Yet the total number of waterfowl (including Common Loons) breeding in these extensive acid-sensitive areas is substantial (roughly 105 000 pairs in 97 500 km²). Regardless of the temporal trends for both atmospheric pollutants and waterfowl populations, a considerable segment of the waterfowl resource in Ontario is potentially at risk from the continued influence of acid precipitation on aquatic systems.

A total of 11 species commonly breed in the region. Several dabbling ducks (Blue-winged Teal, Green-winged Teal and Mallard) and the Wood Duck were commonly found in the Killarney and Nipissing blocks, where agricultural influences, combined with extensive swamp habitat, provide a variety of well-buffered wetland types, not unlike situations in southern Ontario. However, waterfowl of particular concern in acidification research are those that rely on aquatic habitats derived through glacial processes and characterized by low acid-neutralizing capabilities. The intergradation between Great Lakes – St. Lawrence and boreal forest regions, along with the added influence of agricultural land use along the North Channel and Lake Nipissing, provides a natural separation in the distribution of typical boreal lake waterfowl from more southerly species, such as Mallard and Wood Duck. Boreal species, including Common Goldeneye, Hooded Merganser, Ring-necked Duck and American Black Duck, regularly co-occurred on plots throughout much of north-eastern Ontario. These species utilized small head-water lakes and wetlands during the nesting period, whereas the piscivorous Common Loon and Common Merganser co-occurred on plots containing larger lakes and river habitat. Such patterns of habitat use differ from those reported by Hunter *et al.* (1986) and Longcore and Stromborg (unpubl.) in eastern Maine, where marshes and other eutrophic systems were found to be more important as waterfowl habitat than oligotrophic head-water lakes. Despite the overall low breeding densities recorded, small, oligotrophic wetlands were found to be extremely important as waterfowl nesting and brood-rearing habitat in our study. It is clear from our work that the conclusion by Hunter *et al.* (1986) that "relatively few waterfowl in north-eastern North America live in the types of aquatic ecosystems that are most vulnerable to acidification" is invalid in relation to the current situation throughout much of central and northern Ontario. As a significant proportion of the breeding waterfowl resource in north-eastern Ontario is associated with aquatic habitats vulnerable to the effects of acidification (i.e., small head-water lakes) (Haines 1981; Kelso and Minns 1982), waterfowl/food-chain studies were concentrated in these habitats and focussed on these species.

2. Patterns of lake acidification

Patterson (1976) and Murphy *et al.* (1984) investigated the importance of environmental factors in regulating duck populations and acknowledged the importance of limnologic considerations in habitat evaluation. Waterfowl reproductive success is influenced by the natural heterogeneity of the aquatic environment and the many interacting physical and chemical factors that influence the productivity and trophic structure of aquatic systems.

Most of the chemical and physical variability within the small head-water lakes in the present study could be explained on the basis of the four major components: acidity, ionic strength, eutrophication and lake morphometry status. Although effects associated with inputs of airborne contaminants are widespread in North America and reflect long-range pollutant transport, the problem of surface water acidification in Canada remains most pronounced in areas adjacent to point sources emitting large amounts of sulphur and nitrogen oxides. North-east of Sudbury in Ontario, it is reported that "the extent of acidification is as great or greater than any recorded elsewhere" (Beamish *et al.* 1976) and can be largely attributed to the local smelting complexes there. The importance of the acidity and ionic strength factors in explaining the variation in the chemical composition of our Wanapitei study lakes demonstrates the impact of the deposition of acidic substances and the potential moderating influences of bedrock and surficial lithology. In a multivariate classification and description of 187 lakes influenced by acid precipitation in north-eastern Ontario (within a 200-km radius of Sudbury), Pitblado *et al.* (1980) also demonstrated the importance of atmospherically conveyed contaminants from the Sudbury smelting complex and the potential moderating influences of lithologically derived buffering species. The principal components derived in their analyses were very similar to those produced in the current study. Variables directly related to smelting activity, particularly high lake sulphate concentrations arising from the direct deposition of SO_2 and sulphur fumigation, were inversely related to distance from Sudbury. The observed pattern was skewed in a south-west–north-east direction, as a result of the prevailing winds of the area. Despite their proximity to Sudbury, lakes north and north-east of Lake Wanapitei have a wide range of chemical conditions. On average, the Alk:SO_4^{2-} ratio was much lower in lakes surveyed at Wanapitei than at Ranger Lake. As a result, increased acidity and elevated metal levels were found in lakes situated on sensitive bedrock at Wanapitei. Yet there was little evidence of extremely elevated levels of certain metals (Cu, Ni and Zn) whose concentrations would be linked to local deposition effects, as was observed in some lakes sampled in the immediate vicinity of Sudbury (Yan and Miller 1984). Lakes situated on resistant bedrock in the Wanapitei area are well buffered, generally nutrient-rich, and comparable with many productive sites (especially VIXEN lakes) surveyed in the Ranger Lake area. The differing nutrient status of lakes, particularly in relation to phosphorus levels, may reflect variation in natural nutrient loadings as influenced by morphometric (i.e., large drainage basin area relative to lake size) and lithological factors. Despite the importance of organic anions in the natural acidity of certain peatland systems in eastern North America (Gorham *et al.* 1984; Kerekes *et al.* 1984), no such contribution was noted in this study.

In spite of lower overall inputs of acids in the Ranger Lake area, surface water acidification threatens many lakes with low to exhausted buffering capacities in this region. The similarity in chemical composition between certain Ranger Lake lakes (GONG) and many Wanapitei lakes (WHS) suggests that these aquatic systems are sensitive and may suffer significant pH/metal stress in the long term, or seasonally as a consequence of acidic spring melt-water inputs.

3. Testing of hypothesis

The capability of any habitat to support breeding waterfowl is a complex function of various ecological parameters, including nest site characteristics, cover for broods from predators, and duckling food requirements. Food is frequently a limiting resource for ducklings and egg-laying females during the breeding season because they are the top predators in the aquatic food-chain (Swanson and Meyer 1973; Reinecke and Owen 1980). We investigated the hypothesis that the adverse effects of acid precipitation on waterfowl would be mediated through changes in the abundance and availability of prey species.

In north-eastern Ontario, the availability of food for many waterfowl species has been influenced by lake acidification. Such acid-induced changes are largely due to the decline in the abundance of fish, which are one of the first groups to be affected by lake acidification (Schofield 1976). In the acid-stressed area, small non-game fish stocks had fewer individuals and fewer species than Ranger Lake. At moderately high pH levels (5.5), most cyprinid species were absent. Below pH 5.5 several non-cyprinid species occurred, in particular yellow perch, which was recorded at Wanapitei only. Whereas fish species richness was primarily a function of lake acidity, both chemical and physical parameters contributed to the gradient of fish communities observed (Bendell and McNicol 1987*b*). Lake and drainage basin morphometry also influenced fish species composition. We concluded that the separation of lakes with and without cyprinids at Ranger Lake was accounted for by the increased availability of stream habitat in lakes downstream, such that cyprinid species were added to the head-water pond species assemblage as drainage basin size increased.

Adverse effects of lake acidification on small non-game fish species appear to have influenced the breeding capabilities of fish-eating waterbirds in the Wanapitei area. Both Common Loons and Common Mergansers preferred the larger, well-drained oligotrophic lakes that also support diverse cyprinid populations (Bendell and McNicol 1987*b*), as well as brook trout (*Salvelinus fontinalis*) in Ontario (Kelso and Minns 1982). A similar relationship was found in Quebec (DesGranges and Darveau 1985). In the present study, both piscivores were equally abundant in the stressed and unstressed areas during the nest initiation period, occupying nearly one of three lakes surveyed. Yet a much smaller percentage of lakes supported broods at Wanapitei than at Ranger Lake. Although predation may set the general mortality level of waterfowl in any given area, the particularly small percentage of piscivore broods observed in the Wanapitei area suggests that another factor, such as a declining food resource, is also causing nest failure or brood mortality.

Evidence from Ontario (Alvo 1985) and Scandinavia (Almer *et al.* 1978) indicates that loss of fish populations from acidic lakes may be responsible for the observed decline of loons in Ontario and of both loons and mergansers in Sweden. Alvo (1985) reported that loon productivity in lakes less than 100 ha in size within a 100-km radius of Sudbury was negatively correlated with lake acidity.

Our results support the conclusions of Eriksson *et al.* (1980) that many of the biological changes in macroinvertebrate assemblages that coincide with acidification are related to changes in predator–prey relations. The shift to an invertebrate-dominated trophic system, following the disappearance of fish from acidic lakes at Wanapitei, has been characterized by the increased occurrence and abundance of large, free-swimming organisms, such as notonectids, corixids, *Graphoderus liberus* and *Chaoborus americanus*. The same assemblage was found in non-acidic lakes in the Ranger Lake area, where the absence of fish is likely a function of biogeographic isolation or winter anoxia.

Invertebrates eaten by waterfowl are often also exploited by fish, some of which are very efficient competitors of waterfowl (Andersson 1981). The increase in abundance of large predatory insects following a decline in fish stocks in acidified lakes increases the availability of food for species whose young require large amounts of protein-rich insect food for growth. Both Eriksson (1979) and Eadie and Keast (1982) argued that competition between fish and Common Goldeneyes affected the selection of feeding localities and brood-rearing sites for this species. Following the experimental removal of fish, Eriksson (1976) found that goldeneye ducklings fed largely on free-swimming aquatic insects, such as notonectids, corixids and dytiscids, in preference to benthic prey. In the present study, the absence of fish was the single most important factor influencing the suitability of lakes for Common Goldeneyes during the brood-rearing period and was also important for Hooded Mergansers. Ring-necked Ducks and Black Ducks were unaffected by either acidity or fish occurrence in their selection of lakes. Instead, broods of these species occupied shallow, nutrient-rich wetlands, with a well-developed shore-zone community, as observed in southern Quebec (Des-Granges and Darveau 1985).

Ducklings of several species of dabbling and diving ducks feed largely on invertebrate prey (Bartonek 1972; Swanson *et al.* 1979; Reinecke and Owen 1980). In this study, near-shore feeding by young dabblers, such as Black Ducks and Mallards, on aquatic insects was largely unaffected by fish predation, possibly because of the cover afforded by the structurally complex littoral zone of eutrophic lakes. In contrast, the large, mobile prey that expand into the pelagic parts of the lake, following release from fish predation, can be exploited by "pursuit divers," such as Common Goldeneyes and Hooded Mergansers, from a very early age. Because of bill morphology or delays in developing full diving capabilities, the young of other diving ducks, such as Ring-necked Ducks, may not be able to exploit increased stocks of nektonic prey in fishless lakes until a much later stage of development. Downy young of Ring-necked Ducks rely on surface feeding in near-shore vegetation to a much greater extent than Goldeneyes (McNicol, unpubl.).

In summary, comparisons of waterfowl brood production between two areas receiving different rates of acid loading have shown that fewer waterfowl broods are produced in the area of higher deposition in relation to the number of pairs nesting in the spring. The worst case was found for fish-eating species (Common Loon and Common Merganser) in the Wanapitei area, in which five to eight times fewer young were produced. Such piscivorous species are, therefore, immediately at risk from the effects of acidification on fish throughout much of north-eastern Ontario. Similar, though less dramatic, reductions in brood production were found for other waterfowl in this area. Insectivores (Common Goldeneye and Hooded Merganser) may derive short-term benefits from reduced competition with fish for common nektonic insect prey. Omnivores derived no immediate advantage from the disappearance of fish. Whereas most waterfowl select more productive and buffered wetlands to raise their young, small head-water lakes comprise a large proportion of the nesting territories occupied by breeding pairs in some areas and for some species. Our findings indicate that the success of these birds may be lower in areas receiving deposition in the order of 20–30 kg $SO_4^{2-} \cdot ha^{-1} \cdot yr^{-1}$. More information is required to determine whether acidity is limiting the abundance of both fish and invertebrate prey to the detriment of all waterfowl.

Appendices

Appendix 1
Sensitivity of bedrock and derived soils to acid precipitation in the Ranger Lake and Wanapitei study areas

Region	Bedrock sensitivity rating	Bedrock assemblage	Bedrock lithology	Surficial geology
Ranger Lake	High sensitivity: low to insignificant buffering capacity, mainly by cation exchange within clay and silt-sized detritus	Felsic intrusive and metamorphic rocks. Granitoid intrusive rocks and volcanic equivalents	Granitic and gneissic rocks of plutonic origin	GONG: Ground moraine; silty to sandy till VIXEN: (*a*) End moraine; sand, gravel, boulders (*b*) Lacustrine; varved or massive clay, silt, fine sand, sand (*c*) Outwash; sand, fine sand, gravel
Wanapitei				
Low sensitivity (WLS)	Low–intermediate sensitivity: low to high buffering capacity mainly by carbonate anion and by cation exchange in clay and silt-sized detritus	Quirke Lake Group — Carbonate-rich siliceous sedimentary rocks	Calcareous fine- to coarse-grained clastic sedimentary rocks	End moraine; sand, gravel, boulders
Medium sensitivity (WMS)	High–intermediate sensitivity: low to high buffering capacity, mainly cation exchange in clay and silt-sized detritus	Cobalt Group — Non-calcareous siliceous sedimentary rocks. Mafic to intermediate metavolcanics — basaltic and associated sedimentary and volcanic rocks	Low-grade and unmetamorphosed sedimentary rocks. Mafic and felsic metavolcanic and associated metasedimentary rocks, undifferentiated	Ground moraine; silty to sandy till
High sensitivity (WHS)	High sensitivity: low to insignificant buffering capacity mainly by cation exchange within clay and silt-sized detritus	Cobalt Group — Quartzose sandstone. Felsic intrusive and metamorphic rocks — granitoid intrusive rocks and volcanic equivalents	Quartzose sandstone, orthoquartzite. Granitic and gneissic rocks of plutonic origin	Ground moraine; silty to sandy till

Appendix 2
Summary of counts and measurements of lakes by watershed unit and size class for major watersheds in the present study (modified from Cox 1978)

		Lake size classes, ha										Summary statistics				
		>1000		100–999		10–99		1–9.9		<1		Totals		Averages		No. of small lakes (<9.9 ha) per 100 km²
Watershed code		No.	Area, ha	No.	Area, ha	No.	Area, ha	No.	Area, ha	No.	Area, ha	No.	Area, ha	Lake area, ha	Watershed area, ha	
Ranger Lake																
2BF-3	Goulais*	–	–	17	2 966	218	5 680	868	3589	264	149	1367	12 384	8.9	197 896	57.2
2CA-2	Garden*	1	2254	5	1 616	115	2 493	471	1860	135	82	727	8 304	11.3	106 233	57.0
2CB-2	Aubinadong*	–	–	14	3 804	232	5 419	630	2665	216	117	1092	12 005	10.9	166 329	50.9
Totals		1	2254	36	8 386	565	13 592	1969	8114	615	348	3186	32 693	10.4	470 458	54.9
Wanapitei																
2DA-2	Upper Wan.*	2	921	13	1 813	75	1 858	341	1190	138	77	569	5 861	10.1	87 819	54.5
2DA-3	East Wan.*	–	–	4	556	23	658	94	428	11	8	132	1 650	12.1	20 235	51.9
2DA-4	Parkin†	–	–	2	855	27	752	92	385	26	15	147	2 007	13.8	19 433	60.7
2DC-1	Sturgeon*	–	–	23	5 193	172	4 589	595	2361	187	98	977	12 241	12.6	198 098	39.5
2DC-2	Chiniguichi*	4	6071	10	2 646	93	2 439	195	832	25	18	327	12 500	36.8	69 891	31.5
Totals		6	6992	52	11 063	390	10 296	1317	5196	387	216	2152	33 764	17.1	395 476	43.1

* Denotes rivers.
† Denotes creek.

Appendix 3

Summary of sampling methods and results of pilot studies conducted in the Ranger Lake area between 1980 and 1982

Pilot studies to examine waterfowl productivity, water chemistry and food-chain interactions were conducted in the Ranger Lake area between 1980 and 1982, but were most intensive in 1981. Summarized below is a description of those sampling methodologies used only in the pilot studies and the major findings of these studies, specifically as they pertain to the comparative studies implemented in 1983.

1. Methods

The number of lakes surveyed and the parameters measured are outlined in Table I. Several physiographic features of the lakes were measured, including bathymetry, water quality and shoreline vegetation. Water quality data collected in 1981 did not include measures of nutrients or water colour. Food-chain interactions were examined by sampling minnows on 37 lakes and macroinvertebrates on 10 of these. The sweep net sampling procedure followed in 1981 differed from that used in 1983 in that a 700-cm^2 circular net was used and samples were taken at randomly selected sites on each lake during two sampling periods, one in June and the other in July. The minnow trapping procedure was also similar to that followed in 1983, but differed primarily in that the number of trap nights per lake varied from two to seven. A summary of sampling methodologies not described in Methods, Section 2, follows.

1.1. Lake bathymetry

Approximations of littoral zone volume (defined as the volume to a maximum depth of 1.3 m), as well as total lake volume and maximum depth, were calculated from the bathymetric data collected on 50 lakes. Bathymetric contour maps were generated from depth sounding transects on each lake. Depths (in feet) were recorded at 5-s intervals along the transects using depth sounding equipment (Lowrance Bluewater Model LFG-225).

1.2. Water quality

The water sampling programme was designed to determine the major chemical characteristics of lakes selected for waterfowl productivity and food-chain studies. Four sampling series were completed, including September 1980 (N = 50 lakes), April 1981 (N = 55), May 1981 (N = 148) and July 1981 (N = 80.) Although the May 1981 survey covered 148 lakes, information from only 100 lakes is used.

For comparative purposes, only those lakes sampled during all four time periods, including September (fall overturn), early April (spring run-off), late May (during thermal stratification) and late July (following thermal stratification), have been included in the analysis of seasonal fluctuations in chemical parameters. Immediately following peak run-off in April 1981, spring melt-water conditions were examined in 14 lakes located in the VIXEN study area. Mid-lake temperature profiles were taken to differentiate between the surface lens of colder (0–2°C), less dense melt-water which overrides the warmer (4°C), denser endemic water. Surface (0–1 m) and deep (> 1 m) water samples were collected using flexible Tygon tubing.

1.3. Ekman sampling

A total of 30 Ekman dredge samples was collected on each lake. This included 10 samples taken on each of three occasions during June and July 1981. Samples were sieved through 2-mm-diameter wire mesh. The relatively large sieve screen size was chosen to sample large organisms likely to be duck food. Substrates of most head-water lakes in the area were of soft organic muck.

1.4. Emergence trap sampling

Two emergence traps were placed on each of nine lakes, and one placed on the tenth (044R). These enclosures covered a 1-m^2 area of the water's surface and resembled an asymmetrical pyramid with a vertical front. The apex of the structure was darkened with an asphalt felt lining and contained a hole into which was inserted a translucent white plumbing elbow attached to a 500-mL Nalgene bottle. Emerging insects were attracted to the light spot created in the darkened apex of the trap and were preserved in the clearing bottle containing a 2:1 mixture of 95% ethanol and ethylene glycol. Clearing bottles were changed every five to nine days and were open continually from 23–25 June to 22–27 July 1981.

1.5. Shoreline vegetation

Ten littoral quadrats (1 m^2) were selected randomly in the littoral zone (0–1 m deep) of 34 lakes. Three quadrats were located at the water's edge, three at 1 m deep and four at 0.5 m. Each quadrat contained three vertical components: above the surface, at the surface and below the surface. Vegetation within each vertical stratum was identified and percentage cover estimates were made.

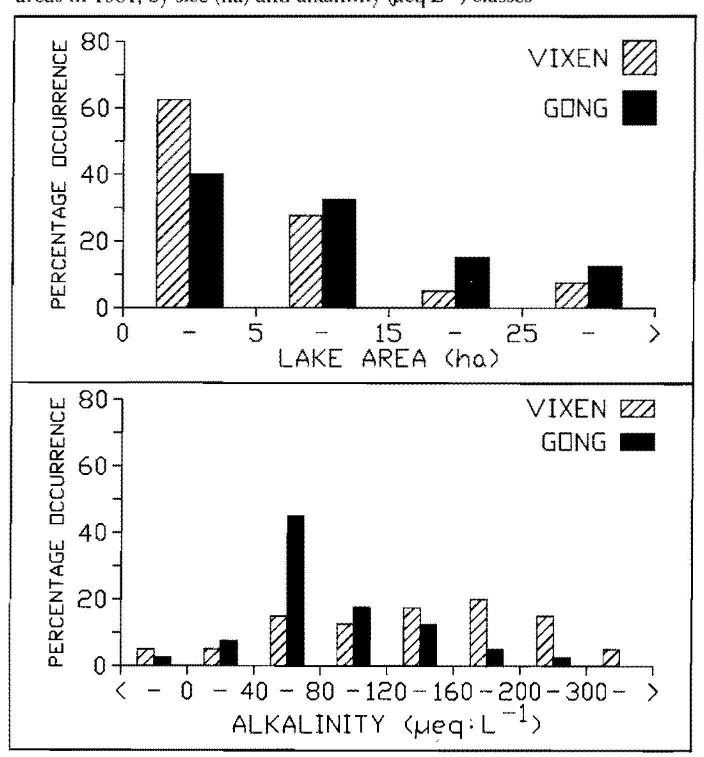
1.6. Waterfowl surveys

Observations on waterfowl reproductive success were made during both 1980 and 1981. Regular ground surveys to assess adult and brood activity were made from mid-May to late July in both years. Ground surveys were expanded in 1981 to encompass 148 lakes; fewer than one-half of these were visited in 1980. In 1981, complete canoe and/or ground checks were made on a total of 100 lakes every 7–10 days, with an additional 48 lakes surveyed less frequently (14–18 days).

Table I
Components of pilot studies conducted in the Ranger Lake area between 1980 and 1982

Study component	Lakes surveyed, N	Method description
Lake physiography		
Morphometry	100	Methods, section 2.2.
Bathymetry	50	Appendix 3
Water quality	148	Methods, section 2.1.
Food-chain studies		Appendix 3
Minnow trap samples	37	Methods, section 2.3.
Sweep net samples	10	Methods, section 2.4.
Ekman samples	10	Appendix 3
Emergence trap samples	10	Appendix 3
Shoreline vegetation	34	Appendix 3
Waterfowl surveys		
Aerial surveys	649	Methods, section 1.
Brood surveys	100	Appendix 3

Figure I
The frequency of occurrence of lakes in the VIXEN and GONG study areas in 1981, by size (ha) and alkalinity (μeq·L^{-1}) classes



Using the Gollop and Marshall (1954) age classification scheme, average dates of the appearance of newly hatched ducklings on brood-rearing lakes were estimated from field observations. Using this information, two 30-day time intervals (28 May – 26 June and 27 June – 26 July) were chosen to represent major phases in the breeding chronology of most species. Adult and brood occupancy estimates were then computed from survey data collected in both 30-day intervals. Only lakes surveyed at least once in both periods were included in the analysis.

2. Results

2.1. Lake physiography and water quality

Lakes surveyed in the Ranger Lake area in 1981 ranged in size from 1 to 73 ha (mean = 10.1 ha), although more than 80% were less than 15 ha in size (Fig. I). More than 57% of the lakes surveyed were either first- or second-order drainage situations and were generally shallow (mean depth = 2.3 m). Only minor differences in lake morphometry were found between VIXEN and GONG study lakes.

Chemical properties for 100 lakes sampled in May 1981 were examined to assess spatial variability in major ion chemistry. While a broad range of water quality was observed, few acidic lakes (pH < 5.5) were found, and there was little evidence of elevated trace metal concentrations. The majority of lakes exhibited a low acid-neutralizing capacity (<200 μeq·L^{-1} alkalinity), and were sensitive to acidification. Substantial differences in the relative sensitivities of lakes were evident, with fewer than 34% of the VIXEN lakes having alkalinities below 120 μeq·L^{-1} compared with 76% of the GONG lakes (Fig. I).

Seasonal differences in water quality between the VIXEN and GONG study areas were evident during all sampling periods (Fig. II), with GONG lakes exhibiting lower average values of pH, conductivity and alkalinity than VIXEN Lakes. Lakes sampled during the spring run-off period (5–9 April) in 1981 had lower average pH, conductivity and alkalinity than at other times of the year (Fig. II). On the 14 lakes where surface (0–1 m) and deep (> 1 m) water samples were collected, lake pH, conductivity, alkalinity and calcium levels were significantly lower in the colder, surface melt-water layer than in the warmer, deeper endemic waters (Table II). During the spring snowmelt of 1981, Kelso *et al.* (1986*b*) also found that head-water lakes (N = 30) sampled in the Algoma District underwent serious declines in alkalinity. Generally, SO_4^{2-}, alkalinity, Ca^{2+} and Mg^{2+} concentrations were reduced by run-off and rain, with lower alkalinity systems showing the greatest changes. This lowering of acid-neutralizing capacity in surface waters suggests that melt-water undergoes little geochemical modification during run-off, even in the naturally well-buffered watersheds of the VIXEN area (App. 1). The accumulation of pollutants in the snowpack and their subsequent release as acidic melt-water is especially critical for shallow head-water systems. In Ontario, snowmelt is the single most important hydrologic event, with between 50 and 70% of the annual run-off occurring in the spring months of March to May. The snowpack melt in the spring can provide between 36 and 77% of the annual acid export from a watershed to a lake (Jeffries *et al.* 1979). Biota inhabiting shallow, near-shore areas of these head-water lakes could be at risk because of the effects of the "acid shock" phenomenon.

Although few acidic lakes were recorded at Ranger Lake, differences in sensitivity among study lakes and between study areas were established. Such sensitive head-water systems might suffer adverse biological effects under current deposition levels, or as a response to snowpack melt.

2.2. Biological studies

2.2.1. *Minnow trap sampling* — Fifteen fish species were recorded from minnow trap sampling on 28 lakes (Fig. III). On the remaining 9 lakes, no fish were recorded. Eight fish taxa were common; these included six cyprinidae (*Chrosomus* spp., pearl dace, common shiner, creek chub, fathead minnow and blacknose dace), brook stickleback and white sucker.

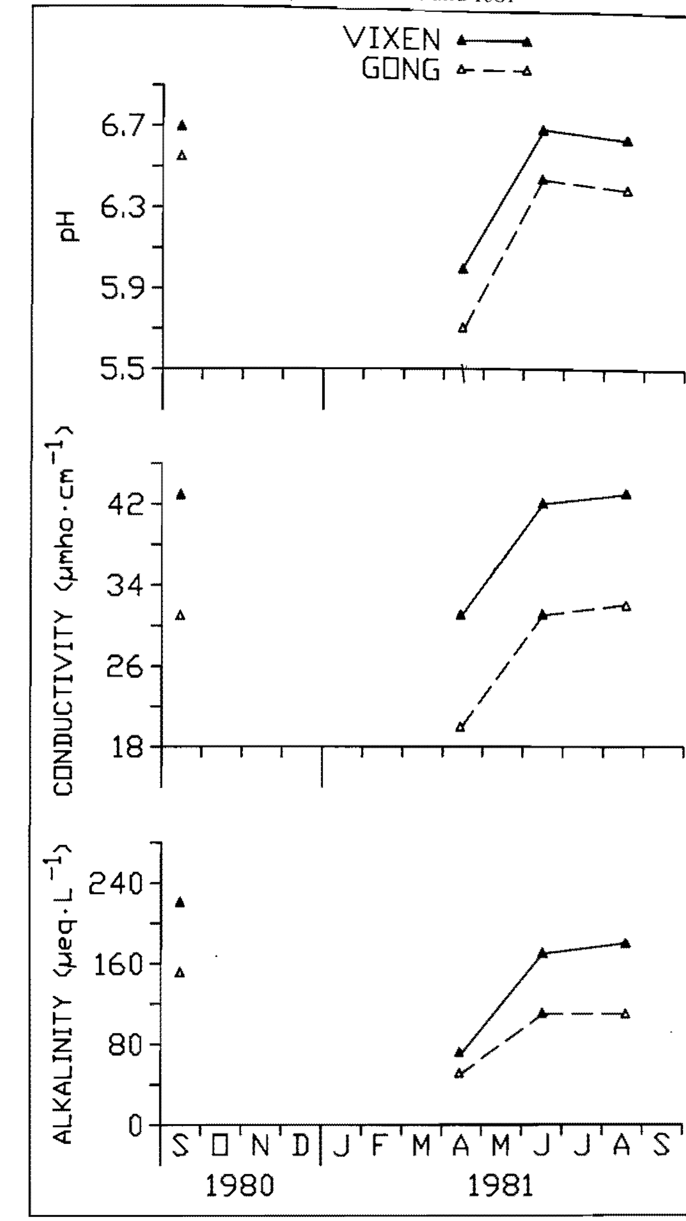
In Figure III, these common taxa have been arranged in sequence, from those occurring in the most lakes to those occurring in only one or two. This arrangement of the species suggests a natural three-way classification of lakes and species. One group of lakes supported only *Chrosomus* spp. and/or pearl dace. Such lakes occurred in both the VIXEN and GONG study areas. The remaining lakes in the GONG area added at least three species from a group containing common shiner, creek chub, white sucker and brook stickleback. In the VIXEN area, lakes added either blacknose dace or fathead minnow. Blacknose dace never occurred in the GONG area, whereas the fathead minnow occurred in small numbers in only one lake in that area. That is, lakes were separated among those containing only *Chrosomus* spp. and/or pearl dace, and those with more diverse assemblages, which differed between the GONG and VIXEN areas.

This classification of lakes conforms to differences in the chemical and physical characteristics of the lakes (Table III). *Chrosomus* spp. and pearl dace are often associated with small, dystrophic bog ponds (Scott and Crossman 1973). In this study, the lakes on which they occurred alone were characterized by small drainage basins. Drainage area size appears to be related to the availability of stream habitat for breeding by several species, including common shiner, creek chub, blacknose dace and white sucker. Streams are also refuges from winter anoxia (Tonn and Magnuson 1982). Lakes in the VIXEN and GONG areas differed in basic chemical characteristics, with lake pH (6.92 vs. 6.44) and alkalinity (171.8 vs. 87.4 μeq·L^{-1}) values significantly higher in the VIXEN area than the GONG area (t-tests, p < 0.01). Both the fathead minnow and blacknose dace were uncommon in GONG lakes, suggesting that the distributions of these species may be restricted by chemical factors. Fathead minnows, in particular, are intolerant up to moderate acidity levels (pH < 6.0) (Schindler and Turner 1982; Rahel and Magnuson 1983; Zischke *et al.* 1983; Mills 1984) and metal levels (Mount 1973).

Table II
Differences of selected chemical parameters (mean ± SD) between surface (0–1 m) and deep (> 1 m) water samples collected on 14 lakes in the Ranger Lake area during the spring run-off period in 1981, tested using paired t-tests (*** p < 0.001, ** < 0.01)

Sample depth	pH (mean ± SD)	Conductivity, μmho·cm^{-1} (mean ± SD)	Alkalinity, μeq·L^{-1} (mean ± SD)	Calcium, mg·L^{-1} (mean ± SD)
Surface (0–1 m)	5.96 ± 0.55	29.3 ± 9.8	81.9 ± 89.5	3.11 ± 1.29
Deep (>1 m)	6.35 ± 0.42	43.0 ± 12.7	209.8 ± 159.3	5.77 ± 1.79
	***	***	**	***

Figure II
Comparison of seasonal and spatial differences in mean lake pH, conductivity (μmho·cm^{-1}) and alkalinity (μeq·L^{-1}) of samples collected in the VIXEN and GONG study areas in 1980 and 1981



2.2.2. *Aquatic invertebrate sampling* — The number of samples from which each major invertebrate taxon was recorded in the total of 20 sweep net samples and 30 Ekman samples from each of the 10 core lakes is shown in Tables IV and V, respectively. Major taxa were defined as those occurring in more than three samples from all lakes. Emergence trap samples were scored for presence–absence of invertebrate taxa only (Table VI).

The sweep net data (Table IV) were analysed using clustering techniques. Average distance dissimilarity values were calculated (Sneath and Sokal 1973) and clustered using Ward's agglomerative method (Wishart 1969). The resulting dendrogram (Fig. IV) indicates a major division between two groups of lakes. One group contains four lakes, of which three (013R, 015R, 459R) had no fish, and the fourth (044R) had so few fish (three minnows collected in 112 h of trapping) that it cannot be considered to have a viable fish population. The remaining group of six lakes contained from two to six fish species (Fig. III).

To identify those invertebrate taxa associated with the two groups of lakes, we performed Mann–Whitney U tests on the scores for each taxa. Ten taxa in sweep net samples were associated with lakes without viable fish populations, as indicated in Table IV. They were the dipteran, *Chaoborus americanus*, backswimmers, *Notonecta* and *Buenoa*, immature corixids, the trichopteran genera, *Oecetis* and *Triaenodes*, the whirligig beetle, *Gyrinus pectoralis*, the diving beetle, *Graphoderus*, the dragonfly, *Leucorrhinia*, and the damselfly, *Lestes*. In marked contrast, only two taxa in Ekman samples, the larvae of the biting midges Ceratopogonidae, and the trichopteran *Oecetis*, were associated with this group of lakes (Table V).

The four lakes without viable fish populations had consistently more invertebrates and a larger number of taxa in sweep net samples (Mann–Whitney U tests, $p = 0.005$). The number of taxa and log number of individuals in sweep net samples tended to fall in a linear fashion with an increase in the number of fish species in each lake (Fig. V).

The number of taxa taken in Ekman samples and the number of individuals taken in both Ekman and emergence trap sampling also tended to be higher on the lakes without viable fish populations, although the differences were less dramatic than for the limnetic fauna sampled using the sweep net procedure.

2.2.3. *Littoral vegetation* — The major aquatic macrophyte taxa recorded in the 34 lakes sampled are summarized in Table VII. A further comparison of the macrophyte community was made among the groups of lakes classified on the basis of fish assemblages (Table VIII). Significant differences were found in the number of taxa and the number of quadrats occupied by floating-leaved and submerged vegetation, and the percentage cover by floating-leaved vegetation. A comparison between aquatic plant and fish species richness suggests that as the number of aquatic plant taxa increases, so does fish species richness ($R^2 = 0.573$, $p < 0.001$) (Fig. VI). In particular, fishless lakes and *Chrosomus* spp. and/or pearl dace lakes usually supported fewer macrophyte taxa and very little floating-leaved vegetation, in terms of either the number of quadrats occupied or percentage cover, compared with either VIXEN or GONG lakes. Both *Nymphaea* sp. and *Potamogeton* spp. showed measurable differences in their occurrence among fish assemblage types, occurring infrequently in fishless lakes (Table VII). This absence of macrophytes, in general, and floating-leaved forms, in particular, in lakes also lacking fish may reflect the same biogeographic factors. That is, the geographic isolation of headwater lakes restricts the access of both fish and macrophytes. Aiken and Gillett (1974) found that the distribution of aquatic plants in Gatineau Park, Quebec, was a function of the isolated positions of some lakes relative to others and of the limited accidental transfer of aquatic plant material to lakes at higher elevations. However, the distribution of some aquatic vascular plants has been linked to the chemical properties (in particular alkalinity) of the lakes in which they grow (Hellquist 1980; Fraser and Morton 1983; Fraser *et al.* 1986). Gorham and Gordon (1963), Wile and Miller (1983) and Wile *et al.* (1985) also found flora to be impoverished in acidic lakes that had elevated metal levels.

2.2.4. *Waterfowl surveys* — Seven waterfowl species were commonly found in the Ranger Lake area in 1980 and 1981 and included Common Goldeneye, Hooded Merganser, American Black Duck, Mallard, Ring-necked Duck, Common Merganser and Common Loon. Other species that nest infrequently are Wood Duck (*Aix sponsa*), Blue-winged Teal (*Anas discors*), Green-winged Teal (*Anas crecca*) and Bufflehead (*Bucephala albeola*). Although nearly 170 duck and loon broods were observed in the two years, fewer than 30% were seen more than once. The likelihood of resighting a brood on a given lake is dependent on its mobility, mortality and visibility. Common Goldeneye broods were frequently observed on the same wetlands (75% of broods resighted) in comparison with other species (25% of broods resighted). Overland travel by ducklings of different species, including Common Merganser, Black Duck, Mallard and Common Goldeneye, has been reported during the brood-rearing period (Erskine 1971; Ball *et al.* 1975), and has been correlated with the food supply in different lakes (Eriksson 1978). Clearly, such movements would increase the vulnerability of broods to predation.

Figure III
The distribution, relative abundance (high > 1, low < 1 fish per 24 trap hours) and assemblage classifications of eight common and six uncommon fish species in 28 lakes in the Ranger Lake area in 1981

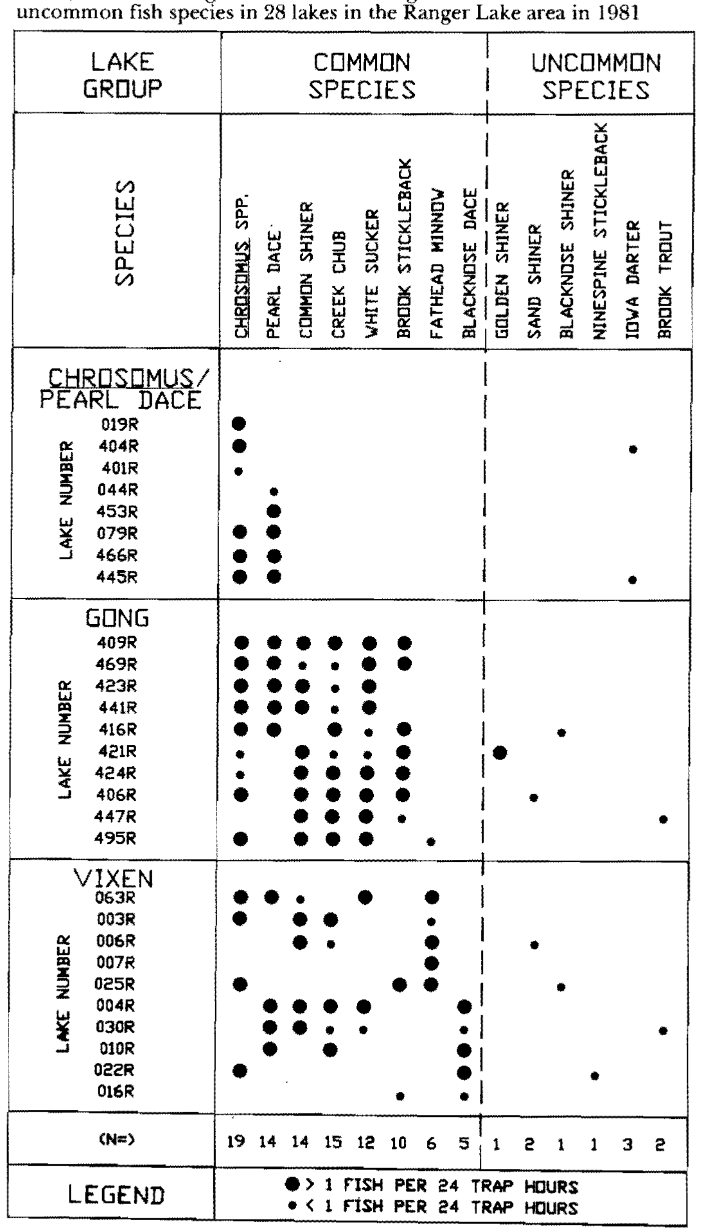
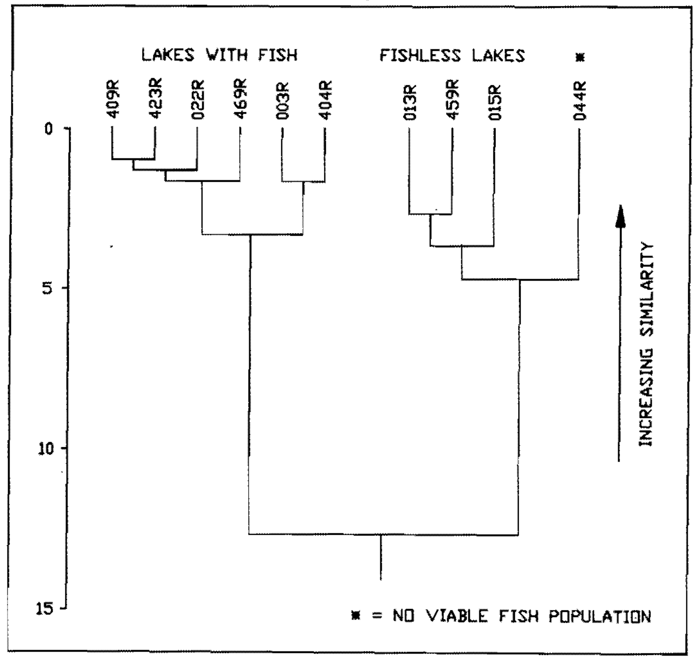


Figure IV
Cluster dendrogram on average distance dissimilarity values, using Ward's method of agglomeration, for aquatic insect taxa in sweep net samples collected in six lakes with fish and four lakes without fish (or a very small fish population) in the Ranger Lake area in 1981



Data from aerial and ground surveys conducted at Ranger Lake were used to assess patterns of habitat selection by waterfowl during the brood-rearing period. Nearly two-thirds of the wetland surveyed supported at least one waterfowl brood in either 1980 or 1981. Whereas information from aerial and brood surveys is limited, significant associations were found between certain species and the occurrence of fish in study lakes (Fig. VII). Aerial surveys of adults (indicated nesting pairs) and ground surveys of both adults and young in 1981 showed that Common Goldeneyes were recorded more often on fishless lakes. The piscivorous species, i.e., Common Merganser and Common Loon, tended to select lakes that contained fish. American Black Duck, Mallard and Ring-necked Duck displayed no obvious preference for either condition, although broods were found more frequently on lakes with fish.

Table III
Comparison of chemical and physical parameters (mean ± SD) for 36 lakes sampled in the Ranger Lake area in 1981, according to fish assemblage types

	Fish assemblage type							
	Fishless* ($N=8$)		*Chrosomus*/ pearl dace ($N=8$)		GONG ($N=10$)		VIXEN ($N=10$)	
Parameter	Mean	± SD	Mean	± SD	Mean	± SD	Mean	± SD
pH	6.41	0.74	6.67	0.32	6.44	0.29	6.92	0.39
Conductivity (μmho·cm^{-1})	34.5	13.5	34.2	12.3	30.0	8.1	41.9	7.2
Alkalinity (μeq·L^{-1})	130.3	133.9	104.7	69.3	87.4	63.5	171.8	80.9
Calcium (mg·L^{-1})	3.88	1.82	3.59	1.31	3.28	1.12	4.23	1.14
Sulphate (mg·L^{-1})	8.08	1.68	7.41	1.38	7.19	0.96	8.28	1.43
Aluminum (μg·L^{-1})	170.4	91.1	60.6	42.8	103.4	26.3	56.3	32.8
Elevation (m)	452	86	465	56	476	30	412	51
Maximum depth (m)	6.4	3.2	8.5	10.3	5.1	4.2	7.7	6.6
Drainage area (ha)	302.9	365.6	98.4	48.0	382.3	282.4	733.0	875.4
Total volume (10 000 m^3)	33.4	49.2	30.9	57.7	32.0	65.9	35.0	54.0
Littoral volume (10 000 m^3)	2.1	3.2	1.3	0.5	3.8	3.1	4.6	4.2
Surface area (ha)	10.6	12.8	6.4	4.7	13.4	14.9	13.4	11.1
Shoreline length (m)	1879	1158	1331	597	2492	1914	2505	1443

* Lake 436R excluded because of incomplete data.

Figure V
Relationship between the number of fish species and the number and log abundance of aquatic invertebrate taxa in sweep net samples collected in 10 lakes in the Ranger Lake area in 1981. The least squares regression line is shown for each comparison

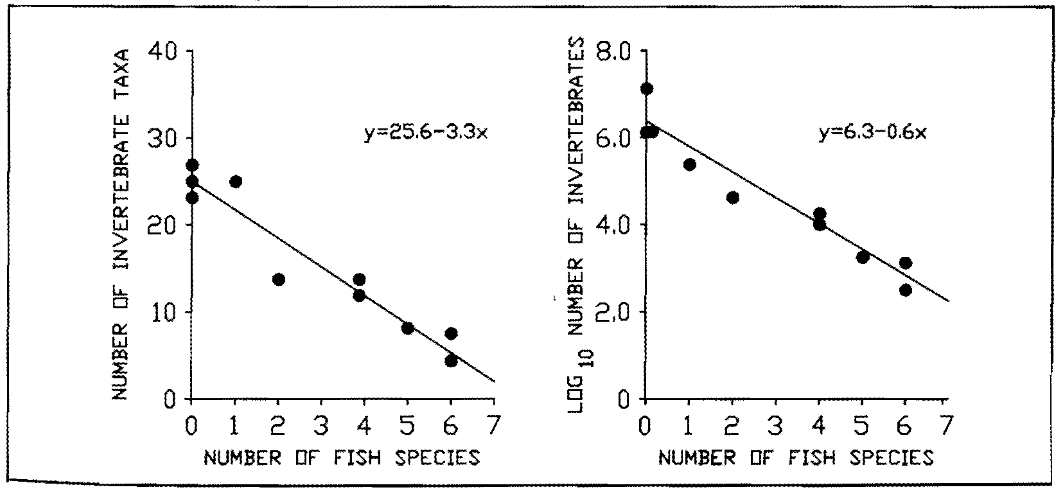


Figure VI
Relationship between the number of fish species and the number of aquatic vascular plant taxa in 34 lakes sampled in the Ranger Lake area in 1981, according to fish assemblage types. The least squares regression line is depicted. Subscripts beside symbols denote multiple observations

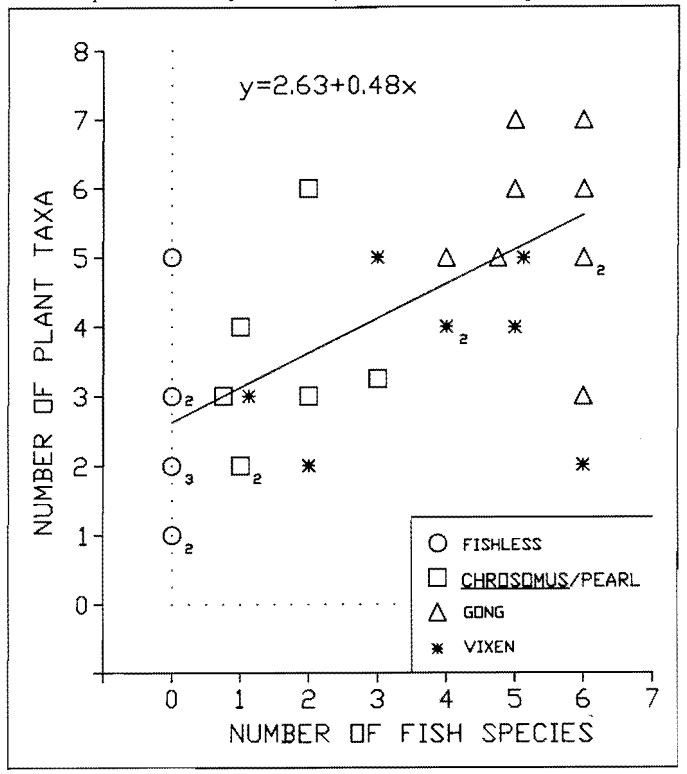


Figure VII
Comparison between lakes with and without fish of indicated nesting pair estimates, adult and/or brood sightings during the brood-rearing period and brood sightings only for lakes surveyed in the Ranger Lake area in 1981 (Fisher exact probability test, ** $p < 0.05$, * $p < 0.10$)

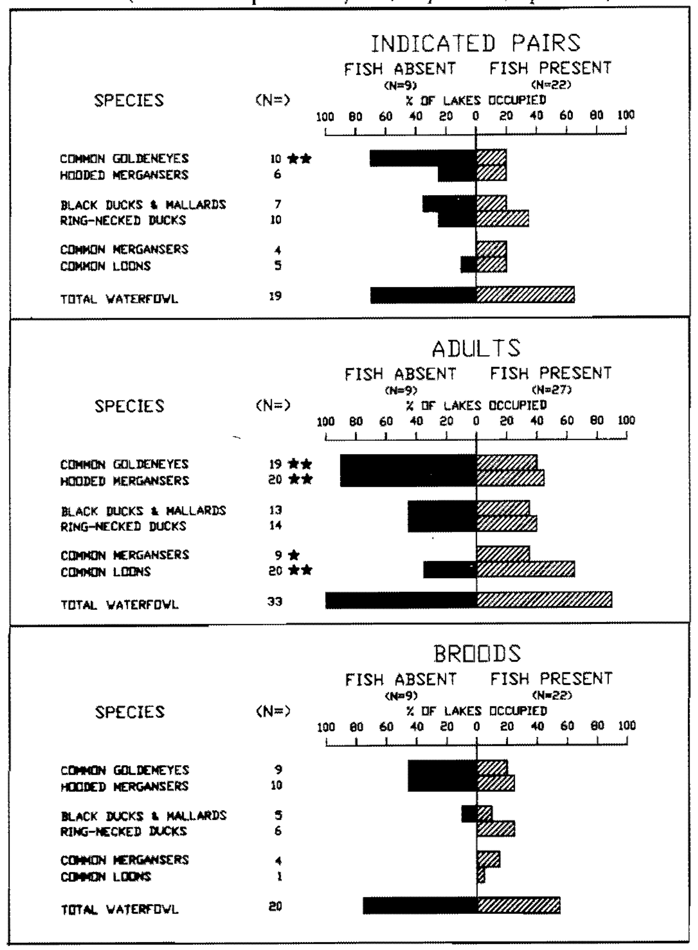


Table IV
The occurrence of aquatic invertebrate taxa in sweep net samples collected in 10 lakes in the Ranger Lake area in 1981, recorded as scores out of 20 samples from each lake†

Taxonomic classification	Lake No. 015R *	013R *	044R *	459R *	003R	022R	404R	409R	423R	469R	*U* test
0. DIPTERA											
1. Chironomidae	14	10	3	9	11	8	9	3	6	2	ns
2. Ceratopogonidae	4	4	–	–	–	1	1	–	–	–	ns
3. *Chaoborus americanus*	6	13	–	8	–	–	–	–	–	–	$p = 0.033$
0. HEMIPTERA											
1. *Buenoa*	1	7	3	1	–	–	–	–	–	–	$p = 0.005$
2. *Notonecta*	12	16	4	13	–	–	–	–	–	–	$p = 0.005$
3. *Sigara* spp. (A)	4	1	7	2	–	2	5	1	–	2	ns
4. Corixidae (I)	19	16	14	18	–	2	1	1	5	4	$p = 0.005$
5. *Gerris*	2	2	–	–	–	2	–	–	–	7	ns
6. *Rheumatobates rileyi*	–	–	1	4	–	1	–	–	–	–	ns
0. TRICHOPTERA											
1. *Mystacides*	–	–	1	–	4	–	–	–	1	–	ns
2. *Oecetis*	1	4	2	1	–	–	1	–	–	–	$p = 0.01$
3. *Triaenodes*	15	1	2	–	–	–	–	–	–	–	$p = 0.033$
4. *Polycentropus*	1	3	1	2	2	–	4	–	–	–	ns
5. *Oxyethira*	4	–	–	1	–	–	–	–	1	–	ns
6. *Limnephilus*	–	1	2	–	–	–	2	–	–	–	ns
7. *Nemotaulius*	–	–	–	–	–	–	3	–	–	–	ns
0. COLEOPTERA											
1. *Gyrinus pectoralis* (A)	–	5	2	3	–	–	–	–	–	–	$p = 0.03$
2. *Gyrinus affinis* (A)	–	1	1	–	–	1	–	–	–	–	ns
3. *Gyrinus* (I)	4	–	–	3	–	–	1	–	1	1	ns
4. *Donacia*	–	–	1	1	–	1	–	–	–	1	ns
5. *Graphoderus*	10	16	3	8	–	–	–	–	–	–	$p = 0.005$
6. *Phyllobius oblongus*	3	–	7	–	–	–	–	–	–	–	ns
0. ODONATA											
1. Aeshnidae	1	3	2	–	1	–	–	–	–	–	ns
2. Coenagrionidae	1	5	–	–	2	1	–	–	1	–	ns
3. *Leucorrhinia*	1	3	–	1	–	–	–	–	–	–	$p = 0.03$
4. *Sympetrum*	1	1	–	–	–	–	1	–	–	–	ns
5. *Lestes*	1	8	5	3	1	–	–	–	–	–	$p = 0.01$
0. EPHEMEROPTERA											
1. Baetidae	–	–	1	–	3	1	–	–	–	–	ns
2. Siphlonuridae	–	–	–	2	1	–	–	–	–	–	ns
AMPHIPODA											
1. *Hyallela azteca*	–	–	6	1	3	–	–	–	–	–	ns
2. *Crangonyx richmondensis*	–	–	5	–	–	–	–	–	–	–	ns
ACARI	12	11	11	10	9	5	13	3	4	–	ns
MOLLUSCA											
1. Sphaeriidae	2	–	1	1	–	–	3	1	–	–	ns
2. *Gyraulus deflectus*	–	–	–	–	2	2	–	–	–	–	ns
3. *Helisoma anceps*	–	–	–	4	2	1	–	–	–	–	ns
OLIGOCHAETA	1	1	1	–	–	1	1	–	–	–	ns
HIRUDINEA	1	–	–	2	–	–	1	–	–	–	ns

† The probability of occurrence (Mann–Whitney *U* statistic) is calculated to test whether a taxon occurred in more samples in lakes without fish (significance level = 0.05; ns = not significant). Lakes without viable fish populations are denoted by an asterisk (*). Adult (A) and immature (I) designations are provided where data for two life stages have not been combined.

Table V
The occurrence of aquatic invertebrate taxa in Ekman dredge samples collected in 10 lakes in the Ranger Lake area in 1981, recorded as scores out of 30 samples from each lake†

Taxonomic classification	Lake No. *015R	*013R	*044R	*459R	003R	022R	404R	409R	423R	469R	U test
0. DIPTERA											
1. Chironomidae	18	25	8	19	20	6	22	9	11	16	ns
2. Ceratopogonidae	3	2	1	5	2	–	1	1	–	–	$p = 0.03$
3. *Chaoborus albatus*	–	–	–	–	–	1	–	–	5	–	ns
4. *Chrysops excitans*	–	–	3	4	1	–	1	–	–	1	ns
0. HEMIPTERA											
1. Corixidae	3	–	–	–	–	–	–	–	–	–	ns
0. TRICHOPTERA											
1. *Polycentropus*	–	–	–	1	2	–	3	–	–	–	ns
2. *Limnephilus*	–	–	1	1	–	–	1	–	–	–	ns
3. *Banksiola*	–	2	–	1	–	–	–	–	–	–	ns
4. *Mystacides*	–	–	3	–	–	–	–	–	–	–	ns
5. *Oecetis*	1	1	2	1	1	–	–	–	–	–	$p = 0.02$
0. COLEOPTERA											
1. *Donacia–Plateumaris*	–	–	–	1	2	1	1	–	–	–	ns
0. ODONATA											
1. *Libellula*	–	–	3	1	2	–	–	–	–	2	ns
2. *Leucorrhinia*	3	7	–	2	–	–	2	–	–	–	ns
3. *Cordulia*	2	13	2	1	3	–	5	1	1	–	ns
4. *Tetragoneuria*	–	–	–	–	3	–	–	–	–	–	ns
5. Gomphidae	1	–	–	–	7	–	–	1	–	1	ns
0. EPHEMEROPTERA											
1. Caenidae	–	–	2	1	6	1	–	2	1	–	ns
2. Siphlonuridae	–	1	–	1	1	–	–	–	–	–	ns
AMPHIPODA											
1. *Hyallela azteca*	–	–	5	3	11	–	–	–	–	–	ns
2. *Crangonyx richmondensis*	1	–	10	1	2	–	1	–	–	–	ns
MOLLUSCA											
1. Sphaeriidae	20	4	8	10	20	2	7	4	2	6	ns
2. *Gyraulus deflectus*	–	–	3	–	–	–	–	–	–	–	ns
OLIGOCHAETA	15	3	–	4	6	–	4	2	1	2	ns
HIRUDINEA	4	–	1	4	1	1	–	2	–	–	ns

† The probability of occurrence (Mann–Whitney U statistic) is calculated to test whether a taxon occurred in more samples in lakes without fish (significance level = 0.05; ns = not significant). Lakes without viable fish populations are denoted by an asterisk (*).

Table VI
The occurrence of aquatic invertebrate taxa collected in individual emergence traps in 10 lakes in the Ranger Lake area during the period 23–25 June to 22–27 July 1981†

Taxonomic classification	Lake No.: *015R	*013R		*044R	*459R		003R		022R		404R		409R		423R		469R	
Trap No.:	1	1	2	1	1	2	1	2	1	2	1	2	1	2	1	2	1	2
0. DIPTERA																		
1. Chironomidae	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1
2. *Chaoborus americanus*	1	1	1	1	–	1	–	–	–	–	–	–	–	–	–	–	–	–
3. *Chaoborus flavicans*	–	–	–	–	–	–	1	–	–	–	1	–	–	–	–	–	1	–
4. *Chaoborus punctipennis*		–	–	–	–	–	–	–	–	–	1	–	1	–	–	1	1	
5. *Chaoborus albatus*	–	–	–	–	–	–	–	–	–	–	–	–	1	1	1	1	–	1
6. *Chaoborus trivittatus*	–	1	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–
7. Culicidae	–	–	–	–	–	–	1	–	–	–	–	–	–	–	–	–	–	–
8. Simuliidae	–	–	–	–	1	–	–	–	–	–	–	1	–	1	–	1	–	–
9. Ceratopogonidae	1	1	–	1	–	–	–	1	1	1	–	–	–	–	–	–	1	–
10. Tipulidae	–	–	–	1	1	–	–	1	–	–	–	–	–	–	–	1	1	–
11. Dolichopodidae	1	–	–	1	–	–	–	–	–	–	–	–	–	–	–	–	–	–
12. Empididae	–	–	–	–	–	1	–	–	1	–	–	–	–	–	–	–	–	–
13. *Chrysops*	–	–	–	–	–	1	–	–	–	–	–	–	–	–	–	–	–	–
14. Ephydridae	1	–	–	–	–	–	–	–	–	–	–	1	–	–	–	–	–	–
0. TRICHOPTERA																		
1. *Triaenodes* sp.	–	1	–	–	1	1	–	–	–	–	1	1	–	–	–	1	1	–
2. *Oecetis* spp.	1	1	–	1	1	–	1	–	–	1	1	1	–	–	1	–	1	1
3. *Mystacides sepulchralis*	–	–	–	–	–	–	–	–	–	1	–	–	–	–	–	–	–	–
4. *Orthotrichia baldulfi*	–	–	–	–	–	1	–	–	–	–	1	1	1	1	1	–	1	1
5. *Oxyethira* sp.	1	–	–	–	1	–	–	–	–	–	–	1	–	–	–	–	–	–
6. *Cheumatopsyche* sp.	1	–	–	–	–	1	–	–	–	1	–	–	–	–	1	–	–	–
7. *Agrypnia vestita*	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	1	–
8. *Agrypnia improba*	1	–	–	1	1	1	1	1	–	1	1	1	–	–	–	–	1	1
9. *Banksiola crotchi*	1	1	1	1	1	1	–	1	–	–	1	1	–	–	–	1	1	–
10. *Phryganea cinerea*	–	–	–	1	–	–	–	–	–	–	–	–	–	–	–	–	–	–
11. *Anabolia bimaculata*	–	–	–	1	1	1	–	–	1	–	–	1	–	–	–	–	–	–
12. *Platycentropus radiatus*	–	1	1	1	1	1	1	–	–	–	1	1	–	1	–	–	1	–
13. *Polycentropus* spp.	1	1	1	1	1	1	1	1	–	–	1	1	1	1	1	1	1	1
14. *Nyctiophylax* sp.	–	–	–	1	–	–	1	–	–	–	1	–	–	–	–	1	–	–
15. *Neureclipsis* sp.	–	1	–	–	–	1	–	–	–	–	–	–	–	–	–	–	–	–
0. ODONATA																		
1. *Enallagma hageni*	–	–	–	–	–	–	–	–	1	–	–	–	–	–	1	1	1	–
2. *Enallagma boreale*	1	1	1	–	–	1	–	–	–	–	–	–	–	–	–	–	–	–
3. *Lestes disjunctus*	1	–	1	1	1	–	–	–	–	–	–	1	–	–	–	–	–	–
4. *Lestes eurinus*	–	–	–	–	–	1	–	–	–	–	–	–	–	–	–	–	–	–
5. Libellulidae	–	–	1	–	–	–	–	–	–	–	–	1	–	–	–	–	–	–
6. Aeshnidae	1	–	–	–	–	1	–	–	–	–	1	–	–	–	–	–	–	–
0. COLEOPTERA																		
1. Curculionidae	–	–	–	1	–	–	–	–	–	–	–	–	–	–	–	1	–	–
2. Chrysomelidae	–	1	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–
3. Helodidae	–	1	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–
0. COLLEMBOLA																		
1. Sminthuridae	–	1	1	–	–	–	1	1	1	–	–	1	1	1	1	1	1	1
2. Isotomidae	–	–	–	–	–	–	–	–	–	–	–	–	1	–	–	–	–	–
0. EPHEMEROPTERA																		
1. Heptageniidae	–	–	–	–	–	–	1	–	–	–	–	–	–	–	–	–	–	–
0. NEUROPTERA																		
1. Sisyridae	1	–	–	–	–	–	1	–	–	–	–	–	–	1	–	–	–	–

† Lakes without viable fish populations are denoted by an asterisk (*). Because of trap disruptions and relocations, data from only one of the two traps on lake 015R are presented. The presence of a taxon has been recorded as 1, whereas blanks (–) denote absence of taxa.

Table VII
Aquatic vascular plants* in 34 lakes in the Ranger Lake area in 1981 scored out of 10 littoral quadrat samples†

Fish assemblage type		No. of quadrats occupied (score out of 10)													
	Aquatic plant taxa:	*Utricularia* (Bladderwort)	*Eriocaulon* (Pipewort)	*Potamogeton* (Pondweed)	*Scirpus* (Bulrush)	*Nymphaea* (White water-lily)	*Sparganium* (Bur reed)	*Eleocharis* (Spike rush)	*Nuphar* (Yellow water-lily)	*Brasenia* (Water shield)	*Myriophyllum* (Water milfoil)	*Dulichium* (3-way sedge)	*Polygonum* (Smartweed)	*Sagittaria* (Arrowhead)	*Equisetum* (Horsetail)
	N:	16	20	18	16	18	10	7	9	4	4	5	3	2	4
Fishless															
422R		–	3	–	–	–	–	–	–	–	–	–	–	–	–
014R		–	–	–	–	–	3	–	–	–	–	–	–	–	–
013R		8	–	–	–	–	2	–	–	–	–	–	–	–	–
420R		8	–	–	–	–	–	–	1	–	–	–	–	–	–
436R		–	3	–	–	–	–	–	2	–	–	–	–	–	–
459R		1	1	–	–	–	3	–	–	–	–	–	–	–	–
015R		–	2	1	3	–	–	–	–	–	–	–	–	–	–
027R		–	5	–	4	–	–	–	–	–	2	1	–	–	1
***Chrosomus*/pearl dace**															
019R		–	3	3	–	–	3	3	–	–	–	–	–	–	–
044R		–	3	–	–	–	–	–	–	–	–	–	–	2	1
401R		–	1	–	–	–	1	–	–	–	–	–	–	–	–
404R		2	–	–	2	2	–	–	–	–	–	–	–	–	–
445R		–	5	–	–	1	1	–	–	–	1	1	–	–	1
453R		–	–	1	1	–	–	–	–	–	–	–	–	–	–
466R		–	1	–	1	1	–	–	–	–	–	–	–	–	–
GONG															
421R		–	–	–	–	1	–	1	1	–	–	–	–	–	–
409R		7	2	–	2	1	–	–	–	–	–	1	–	–	–
416R		–	2	5	–	–	–	1	1	–	–	–	1	–	–
424R		–	–	5	–	1	3	–	4	–	–	1	–	–	–
495R		3	–	2	8	1	–	–	–	–	2	–	–	–	–
441R		8	1	–	4	7	–	1	–	–	–	–	1	–	–
406R		1	–	2	5	5	2	–	–	4	–	–	–	–	–
423R		1	1	1	3	2	–	4	2	–	–	–	–	–	–
469R		5	2	2	2	4	–	–	–	1	–	2	–	–	–
VIXEN															
016R		–	–	3	–	5	–	–	–	–	–	–	–	–	–
030R		–	7	–	–	3	–	–	–	–	–	–	–	–	–
007R		10	–	3	1	–	–	–	–	–	–	–	–	–	–
025R		–	–	3	–	–	–	1	–	–	–	–	1	1	–
006R		4	5	1	–	–	–	1	–	–	–	–	–	–	–
063R		–	–	1	1	3	–	–	–	2	–	–	–	–	–
022R		1	1	3	–	5	–	–	3	–	–	–	–	–	–
004R		6	–	3	5	3	–	–	–	4	–	–	–	–	–
003R		4	1	1	2	7	1	–	2	–	3	–	–	–	–
010R		7	2	1	1	1	1	–	1	–	–	–	–	–	1
Chi-squared statistic		4.75	2.15	12.16	4.38	13.86	1.81	5.10	3.91	3.47	2.52	4.29	3.37	2.24	3.05
	($p<$)	ns	ns	0.007	ns	0.003	ns	ns	ns	ns	ns	ns	ns	ns	ns

* Taxa not included were the graminoids and *Carex* spp., which occurred on almost all lakes, and those that occurred on only one lake, which included a water parsley (Umbelliferae), water lobelia (*Lobelia dortmanna*) and *Littorella* sp.
† Chi-squared tests were performed on each taxon, according to fish assemblage types.

Table VIII
The occurrence and relative abundance (mean percentage cover) of floating-leaved and submerged aquatic vascular plants in 34 lakes in the Ranger Lake area in 1981*

Fish assemblage type	No. of taxa	No. of quadrats occupied (score out of 10)		Mean percentage cover (N = 10 quadrats)	
		Floating-leaved	Submerged	Floating-leaved	Submerged
Fishless					
422R	1	0	3	0	2
014R	1	0	3	0	1
013R	2	0	8	0	39
420R	2	1	8	1	1
436R	2	0	3	5	10
459R	3	0	4	0	3
015R	3	0	5	0	20
027R	5	0	6	0	39
***Chrosomus*/pearl dace**					
019R	4	0	8	0	12
044R	3	1	4	1	7
401R	2	1	1	1	1
404R	3	2	4	1	13
445R	6	2	7	1	18
453R	2	0	2	0	14
466R	3	1	5	1	37
GONG					
421R	3	1	1	1	1
409R	5	1	7	1	5
416R	5	5	6	15	19
424R	5	7	6	16	7
495R	5	4	9	6	39
441R	6	7	8	8	27
406R	6	8	9	24	51
423R	7	4	7	12	27
469R	7	6	7	5	26
VIXEN					
016R	2	5	5	8	3
030R	2	2	8	2	8
007R	3	0	10	0	42
025R	4	0	7	1	27
006R	4	1	8	1	25
063R	4	4	4	8	11
022R	5	8	9	9	2
004R	5	8	8	18	19
003R	8	7	8	9	16
010R	8	3	8	20	20
Chi-squared statistic	13.32	17.23	8.98	16.79	1.89
($p<$)	0.004	0.0006	0.029	0.0008	ns

* A Kruskal–Wallis one-way ANOVA by ranks tested differences in occurrence and relative abundance of each vegetation type among fish assemblage types.

Appendix 4
Summary of the procedures and principal equipment used in water chemical determinations

No.	Chemical determination	Procedure	Principal equipment
1	pH	Ross electrode	Fisher Accumet Model 750 (Specific Ion)
2	Specific conductance	$\mu S \cdot cm^{-1}$ at 25°C	Radiometer Type CDM2e Conductivity Meter
3	Total alkalinity	Electrometric titration	Metrohm E636 Titroprocessor
4	Total nitrogen	Automated micro-Kjeldahl	Technicon Auto-Analyzer II
5	NH_3 – nitrogen	Automated sodium nitroprusside	Technicon Auto-Analyzer IIC plus
6	$NO_2 + NO_3$ – nitrogen	Automated cadmium reduction	Technicon Auto-Analyzer IIC plus
7	Total phosphorus	Automated molybdophosphoric blue	Technicon Auto-Analyzer IIC plus
8	Soluble reactive phosphorus	Automated molybdophosphoric blue	Technicon Auto-Analyzer IIC plus
9	Potassium (K)	Flame emission	Varian 1275 Spectrophotometer
10	Sodium (Na)	Flame emission	Varian 1275 Spectrophotometer
11	Calcium (Ca)	Atomic absorption	Varian 1275 Spectrophotometer
12	Magnesium (Mg)	Atomic absorption	Varian 1275 Spectrophotometer
13	Sulphate (SO_4)	Automated methyl thymol blue	Technicon Auto-Analyzer IIC plus
14	Chloride (Cl)	Automated mercuric thiocyanate	Technicon Auto-Analyzer IIC plus
15	Silica (SiO_2)	Automated ascorbic acid	Technicon Auto-Analyzer IIC plus
16	Iron (Fe)	Atomic absorption	Perkin-Elmer 4000 Spectrophotometer & Varian 975 (Graphite Furnace)
17	Aluminum (Al)	Atomic absorption	Perkin-Elmer 4000 Spectrophotometer & Varian 975 (Graphite Furnace)
18	Manganese (Mn)	Atomic absorption	Perkin-Elmer 4000 Spectrophotometer & Varian 975 (Graphite Furnace)
19	Zinc (Zn)	Atomic absorption	Perkin-Elmer 4000 Spectrophotometer & Varian 975 (Graphite Furnace)
20	Copper (Cu)	Atomic absorption	Perkin-Elmer 4000 Spectrophotometer & Varian 975 (Graphite Furnace)
21	Nickel (Ni)	Atomic absorption	Perkin-Elmer 4000 Spectrophotometer & Varian 975 (Graphite Furnace)
22	Cadmium (Cd)	Atomic absorption	Perkin-Elmer 4000 Spectrophotometer & Varian 975 (Graphite Furnace)
23	Lead (Pb)	Atomic absorption	Perkin-Elmer 4000 Spectrophotometer & Varian 975 (Graphite Furnace)
24	Inorganic carbon (TIC)	UV oxidation	Astro Model 1850 Total Organic Carbon Analyzer
25	Organic carbon (TOC)	UV oxidation	Astro Model 1850 Total Organic Carbon Analyzer
26	Total carbon (TC)	UV oxidation	Astro Model 1850 Total Organic Carbon Analyzer
27	Water colour	Hazen platinum–cobalt scale	Helige Aqua Testa

Appendix 5
Annotated list of waterfowl species in the survey region

1. **Common Loon** *(Gavia immer)* — This widespread and common piscivore is found in large lakes. There is no obvious reason for low counts in the Sault Ste. Marie and Nipissing blocks, although the density of small lakes (<9.9 ha) is much higher in the Eastern Lake Superior Tributary Watershed (2B) (Fig. 3*d*) than elsewhere in the region. This species was often found on known acidic lakes.

2. **Canada Goose** *(Branta canadensis)* — A small resident flock breeds around Pumpkin Point (Sault Ste. Marie block) where a plot happened to fall.

3. **Mallard** *(Anas platyrhynchos)* — An uncommon species in the northern tier of blocks, but much more common to the south, particularly around agricultural areas. This species has risen rapidly in numbers in southern Ontario (Ross *et al.* 1984), but no such increase is as yet evident in the north.

4. **American Black Duck** *(Anas rubripes)* — This species is common throughout the region. The count in the Huntsville block is the highest so far recorded for such an area anywhere in Ontario. Although the overall population of the Black Duck is currently declining at a slow but steady rate, no such decline is evident from any survey data for this region. The species regularly co-occurs with the Ring-necked Duck (Ross 1987) and is found in a wide range of smaller wetlands, particularly those that are shallow with well-developed littoral zones. This is often the only species found on peat-dominated wetlands (bogs and fens) (Blancher and McNicol 1986).

5. **Northern Pintail** *(Anas acuta)* — A single male was noted.

6. **Green-winged Teal** *(Anas crecca)* — Generally found at low densities and sporadically distributed in the region; higher counts were associated with swampy habitat, particularly in the Nipissing block, but also in the Killarney and Huntsville blocks.

7. **Blue-winged Teal** *(Anas discors)* — Similar in distribution and abundance to the previous species, the Blue-winged Teal was most often encountered on larger open marshes.

8. **Wood Duck** *(Aix sponsa)* — All evidence indicates that this currently uncommon species is increasing rapidly in the southern part of the region, as it did throughout southern Ontario (Ross *et al.* 1984). Birds were also occasionally reported in the northern blocks. It was usually found in or near flooded forests.

9. **Ring-necked Duck** *(Aythya collaris)* — This was the most abundant and widespread species, being well represented in all blocks. Although present in a wide range of lake types, it was often found on lakes with extensive sections of well-developed littoral zone. As this species nests in mid-June, one must be careful in making conclusions on breeding habitat from the survey results.

10. **Lesser Scaup** *(Aythya affinis)* — Four pairs were recorded near Pumpkin Point and may not necessarily be breeding.

11. **Common Goldeneye** *(Bucephala clangula)* — This species was common only in the northern more boreal plots and appeared to show a decline in abundance from west to east. The species was very rare in the southern blocks. This and the other of the *Mergini* tribe seemed to prefer lakes with deeper waters, often with less littoral zone than those of the previous species (Ross 1987).

12. **Bufflehead** *(Bucephala albeola)* — This rare species breeds sporadically throughout the region.

13. **Hooded Merganser** *(Lophodytes cucullatus)* — This common species was recorded in good numbers in all but the Killarney blocks. The low count in that area may be an artifact, as previous surveys (Table 3) give results comparable with those of the other blocks.

14. **Common Merganser** *(Mergus merganser)* — This species was similar in abundance to the previous species, although more likely to be found on larger lakes and on running water such as rivers and streams.

15. **Red-breasted Merganser** *(Mergus serrator)* — A very rare species usually associated with the largest lakes (e.g., Lake Superior, Lake Nipissing).

Appendix 6*a*
Indicated nesting pair occupancy frequencies for the Common Loon in the Ranger Lake and Wanapitei study areas, in relation to selected habitat parameters

Area of open water, ha	Shoreline development class	Length of well-vegetated littoral zone, m	Ranger Lake		Wanapitei	
			Present	Absent	Present	Absent
0–0.40	1	0–100	0	54	0	6
		101–500	0	13	0	17
	2	0–100	0	25	0	1
		101–500	0	19	0	21
	3	0–100	0	17	–	–
		101–500	0	39	0	4
		>500	0	21	0	6
0.41–1.50	1	0–100	0	24	0	1
		101–500	0	27	0	38
		>500	–	–	0	1
	2	0–100	0	3	–	–
		101–500	0	21	0	13
		>500	0	13	0	19
	3	0–100	0	4	–	–
		101–500	0	5	–	–
		>500	0	25	0	24
1.51–4.00	1	0–100	1	17	–	–
		101–500	1	11	0	4
		>500	0	15	0	28
	2	0–100	0	19	–	–
		101–500	0	8	–	–
		>500	3	19	1	50
	3	0–100	0	7	–	–
		101–500	0	11	–	–
		>500	1	22	1	31
4.01–20.00	1	0–100	4	14	–	–
		101–500	2	3	–	–
		>500	2	10	5	10
	2	0–100	10	19	–	–
		101–500	9	22	–	–
		>500	6	19	18	40
	3	0–100	1	5	–	–
		101–500	1	3	–	–
		>500	4	28	19	35
>20	2	0–100	2	2	–	–
		101–500	–	–	1	0
		>500	1	5	3	6
	3	0–100	3	1	–	–
		>500	3	2	10	1

Appendix 6*b*
Indicated nesting pair occupancy frequencies for combined Mallard/Black Ducks in the Ranger Lake and Wanapitei study areas, in relation to selected habitat parameters

Area of open water, ha	Shoreline development class	Length of well-vegetated littoral zone, m	Ranger Lake		Wanapitei	
			Present	Absent	Present	Absent
0–0.40	1	0–100	0	54	0	6
		101–500	0	13	0	17
	2	0–100	0	25	0	1
		101–500	0	19	2	19
	3	0–100	1	16	–	–
		101–500	2	37	0	4
		>500	0	21	0	6
0.41–1.50	1	0–100	2	22	0	1
		101–500	2	25	3	35
		>500	–	–	0	1
	2	0–100	0	31	–	–
		101–500	3	18	1	12
		>500	1	12	4	15
	3	0–100	0	4	–	–
		101–500	0	5	–	–
		>500	4	21	3	21
1.51–4.00	1	0–100	2	16	–	–
		101–500	3	9	1	3
		>500	4	11	2	26
	2	0–100	4	15	–	–
		101–500	3	5	–	–
		>500	4	18	12	39
	3	0–100	1	6	–	–
		101–500	1	10	–	–
		>500	6	17	7	25
4.01–20.00	1	0–100	2	16	–	–
		101–500	1	4	–	–
		>500	4	8	1	14
	2	0–100	5	24	–	–
		101–500	4	27	–	–
		>500	3	22	13	45
	3	0–100	1	5	–	–
		101–500	0	4	–	–
		>500	8	24	14	40
>20	2	0–100	0	4	–	–
		101–500	–	–	0	1
		>500	0	6	0	9
	3	0–100	0	4	–	–
		>500	1	4	2	9

Appendix 6*c*
Indicated nesting pair occupancy frequencies for the Ring-necked Duck in the Ranger Lake and Wanapitei study areas, in relation to selected habitat parameters

Area of open water, ha	Shoreline development class	Length of well-vegetated littoral zone, m	Ranger Lake		Wanapitei	
			Present	Absent	Present	Absent
0–0.40	1	0–100	1	53	0	6
		101–500	2	11	0	17
	2	0–100	0	25	0	1
		101–500	1	18	1	20
	3	0–100	1	16	–	–
		101–500	0	39	0	4
		>500	1	20	1	5
0.41–1.50	1	0–100	0	24	0	1
		101–500	1	26	2	36
		>500	–	–	0	1
	2	0–100	2	29	–	–
		101–500	3	18	1	12
		>500	2	11	1	18
	3	0–100	0	4	–	–
		101–500	0	5	–	–
		>500	4	21	2	22
1.51–4.00	1	0–100	2	16	–	–
		101–500	2	10	0	4
		>500	4	11	3	25
	2	0–100	3	16	–	–
		101–500	1	7	–	–
		>500	4	18	4	47
	3	0–100	1	6	–	–
		101–500	1	10	–	–
		>500	6	17	5	27
4.01–20.00	1	0–100	0	18	–	–
		101–500	1	4	–	–
		>500	1	11	2	13
	2	0–100	1	28	–	–
		101–500	4	27	–	–
		>500	4	21	8	50
	3	0–100	0	6	–	–
		101–500	0	4	–	–
		>500	9	23	7	47
>20	2	0–100	0	4	–	–
		101–500	–	–	0	1
		>500	0	6	0	9
	3	0–100	0	4	0	0
		>500	0	5	0	11

Appendix 6*d*
Indicated nesting pair occupancy frequencies for the Common Goldeneye in the Ranger Lake and Wanapitei study areas, in relation to selected habitat parameters

Area of open water, ha	Shoreline development class	Length of well-vegetated littoral zone, m	Ranger Lake		Wanapitei	
			Present	Absent	Present	Absent
0–0.40	1	0–100	0	54	0	6
		101–500	0	13	0	17
	2	0–100	0	25	0	1
		101–500	1	18	0	21
	3	0–100	1	16	–	–
		101–500	1	38	0	4
		>500	1	20	0	6
0.41–1.50	1	0–100	5	19	0	1
		101–500	2	25	2	36
		>500	–	–	0	1
	2	0–100	2	29	–	–
		101–500	0	21	2	11
		>500	1	12	0	19
	3	0–100	0	4	–	–
		101–500	0	5	–	–
		>500	1	24	1	23
1.51–4.00	1	0–100	7	11	–	–
		101–500	4	8	0	4
		>500	5	10	8	20
	2	0–100	6	13	–	–
		101–500	3	5	–	–
		>500	5	17	12	39
	3	0–100	1	6	–	–
		101–500	2	9	–	–
		>500	5	18	9	23
4.01–20.00	1	0–100	5	13	–	–
		101–500	0	5	–	–
		>500	3	9	2	13
	2	0–100	5	24	–	–
		101–500	2	29	–	–
		>500	8	17	10	48
	3	0–100	2	4	–	–
		101–500	0	4	–	–
		>500	7	25	9	45
>20	2	0–100	0	4	–	–
		101–500	–	–	0	1
		>500	0	6	1	8
	3	0–100	0	4	–	–
		>500	0	5	2	9

Appendix 6e
Indicated nesting pair occupancy frequencies for the Hooded Merganser in the Ranger Lake and Wanapitei study areas, in relation to selected habitat parameters

Area of open water, ha	Shoreline development class	Length of well-vegetated littoral zone, m	Ranger Lake		Wanapitei	
			Present	Absent	Present	Absent
0–0.40	1	0–100	0	54	0	6
		101–500	0	13	0	17
	2	0–100	0	25	0	1
		101–500	0	19	0	21
	3	0–100	0	17	–	–
		101–500	3	36	1	3
		>500	0	21	0	6
0.41–1.50	1	0–100	1	23	0	1
		101–500	2	25	4	34
		>500	–	–	0	1
	2	0–100	3	28	–	–
		101–500	1	20	1	12
		>500	1	12	3	16
	3	0–100	1	3	–	–
		101–500	0	5	–	–
		>500	0	25	3	21
1.51–4.00	1	0–100	2	16	–	–
		101–500	1	1	1	3
		>500	1	14	5	23
	2	0–100	4	15	–	–
		101–500	0	8	–	–
		>500	1	21	10	41
	3	0–100	0	7	–	–
		101–500	1	10	–	–
		>500	7	16	6	26
4.01–20.00	1	0–100	0	18	–	–
		101–500	0	5	–	–
		>500	2	10	2	13
	2	0–100	2	27	–	–
		101–500	3	28	–	–
		>500	6	19	8	50
	3	0–100	1	5	–	–
		101–500	0	4	–	–
		>500	4	28	7	47
>20	2	0–100	0	4	–	–
		101–500	–	–	0	1
		>500	0	6	0	9
	3	0–100	0	4	–	–
		>500	1	4	0	11

Appendix 6f
Indicated nesting pair occupancy frequencies for the Common Merganser in the Ranger Lake and Wanapitei study areas, in relation to selected habitat parameters

Area of open water, ha	Shoreline development class	Length of well-vegetated littoral zone, m	Ranger Lake		Wanapitei	
			Present	Absent	Present	Absent
0–0.40	1	0–100	0	54	0	6
		101–500	0	13	0	17
	2	0–100	0	25	0	1
		101–500	0	19	0	21
	3	0–100	0	17	–	–
		101–500	0	39	0	4
		>500	1	20	0	6
0.41–1.50	1	0–100	0	24	0	1
		101–500	0	27	1	37
		>500	–	–	0	1
	2	0–100	0	31	0	0
		101–500	0	21	0	13
		>500	0	13	0	19
	3	0–100	0	4	–	–
		101–500	0	5	–	–
		>500	1	24	0	24
1.51–4.00	1	0–100	0	18	–	–
		101–500	0	12	0	4
		>500	1	14	0	28
	2	0–100	2	17	–	–
		101–500	1	7	–	–
		>500	0	22	2	49
	3	0–100	1	6	–	–
		101–500	0	11	–	–
		>500	1	22	3	29
4.01–20.00	1	0–100	0	18	–	–
		101–500	0	5	–	–
		>500	0	12	0	15
	2	0–100	5	24	–	–
		101–500	1	30	–	–
		>500	2	23	4	54
	3	0–100	0	6	–	–
		101–500	0	4	–	–
		>500	2	30	5	49
>20	2	0–100	0	4	–	–
		101–500	–	–	1	0
		>500	2	4	2	7
	3	0–100	1	3	–	–
		>500	0	5	2	9

Appendix 7
Chemical composition of fresh waters in unpolluted and polluted areas of Scandinavia and eastern North America, including median estimates from the present study, according to bedrock sensitivity groups (Ranger Lake—VIXEN and GONG; Wanapitei—high (WHS), moderate (WMS) and low (WLS) sensitivity)†

Region	*N*	Ionic concentrations, $\mu eq \cdot L^{-1}$									Reference
		Ca^{2+}	Mg^{2+}	Na^{+}	K^{+}	Total cations	HCO_3^-	SO_4^{2-}	(Cl^-)	Total anions	
Unpolluted											
Scandinavia											
Sweden 1	47	125	70	27	11	233	208	29	(26)	237	Henriksen 1982
Sweden 2	21	577	136	39	38	790	645	87	(102)	732	Henriksen 1982
Sweden 3	21	248	108	32	17	405	247	73	(64)	320	Henriksen 1982
Sweden 4	65	149	8	60	14	231	250	37	(30)	287	Henriksen 1982
Norway (N)	21	128	60	15	6	209	138	45	(85)	183	Henriksen 1982
Norway (W)	25	37	6	10	3	56	28	29	(40)	57	Henriksen 1982
United States											
Georgia	5	340	44	22	23	429	398	17	(149)	415	Henriksen 1982
Minnesota	58	299	114	23	12	448	313	60	(21)	373	Henriksen 1982
Canada											
NW Territories	9	385	197	17	26	625	460	98	(120)	558	Armstrong & Schindler 1971
Saskatchewan	5	290	260	110	43	703	530	69	–	599	Armstrong & Schindler 1971
Manitoba	17	169	73	8	12	262	182	58	(42)	240	Armstrong & Schindler 1971
Ontario											
•James Bay	–	65	19	20	9	113	84	37	(21)	121	Henriksen 1982
•ELA[1]‡	103	98	49	21	9	177	99	76	(21)	175	Beamish *et al.* 1976
•ELA[2]	40	78	68	6	10	162	62	59	(39)	121	Armstrong & Schindler 1971
Quebec											
•Arctic tundra	8	31	21	10	3	65	80	6	(32)	86	Potvin & Grimard 1983
•Alpine tundra	7	93	118	12	3	226	157	68	(6)	225	Potvin & Grimard 1983
•Taiga	38	124	125	18	5	210	163	206	(5)	369	Potvin & Grimard 1983
•Muskeg	14	59	62	12	3	136	136	27	(6)	163	Potvin & Grimard 1983
Polluted											
Canada											
Ontario											
•Sudbury Area											
–Whitefish Lake	27	323	159	21	18	521	190	404	(56)	594	Beamish & Harvey 1972
–LaCloche Mt.	6	97	31	–	6	134	10	207	(29)	217	Armstrong & Schindler 1971
–Killarney	22	150	75	26	10	261	0	290	?	290	Beamish & Harvey 1972
–Sudbury	4	280	110	63	22	475	0	530	(*)	530	Dillon *et al.* 1980
–Hannah		567	280	87	53	1038	0	1192	(76)	1192	Yan & Miller 1984
–Middle		490	410	104	49	1090	0	927	(*)	927	Yan & Miller 1984
–Lohi		310	205	83	33	672	0	556	(*)	556	Yan & Miller 1984
–Clearwater		283	103	17	17	499	0	543	(62)	543	Yan & Miller 1984
–Labelle		254	88	36	16	395	76	324	(14)	400	Yan & Miller 1984
–Nelson		205	80	32	12	331	40	328	(12)	368	Yan & Miller 1984
–Mountaintop		150	80	32	11	297	0	291	(11)	291	Yan & Miller 1984
–Wanapitei – WLS	11	206	66	28	16	319	52	221	(12)	275	Present study
–Wanapitei – WMS	24	184	55	33	12	276	17	218	(11)	238	Present study
–Wanapitei – WHS	20	101	42	26	11	210	0	236	(9)	236	Present study
–Ranger Lake – GONG	34	133	45	18	13	212	53	142	(15)	192	Present study
–Ranger Lake – VIXEN	32	198	76	29	14	319	141	154	(19)	293	Present study
•Muskoka (1967–68)	5	94	104	15	8	221	67	79	(17)	146	Dillon *et al.* 1980
•Muskoka (1976–79)	14	149	69	21	13	252	110	168	(18)	278	Dillon *et al.* 1980
•Ottawa R. drainage	13	370	170	–	–	540	320	210	(0)	530	Armstrong & Schindler 1971
Quebec											
•Upper Laurentians	10	95	33	31	6	165	52	69	(6)	121	Rodrigue & DesGranges 1986
•Middle Laurentians	62	200	58	28	10	296	140	110	(9)	250	Rodrigue & DesGranges 1986
•Appalachians	8	220	66	28	8	322	152	89	(8)	241	Rodrigue & DesGranges 1986
Nova Scotia											
•Kejimkujik		38	39	135	6	218	0	52	(121)	–	Kerekes 1980
•Beaverskin		19	30	115	7	171	0	31	(116)	31	Kerekes *et al.* 1986
•Pebbleloggitch		19	30	120	7	176	0	47	(113)	47	Kerekes *et al.* 1986
United States											
Adirondacks	38	146	43	16	7	212	74	121	(9)	195	Charles 1985

† All estimates have been adjusted for sea salts with exceptions denoted by an asterisk (*). Chloride (Cl^-) values have been provided where available.

‡ ELA = Experimental Lakes Area, Northwestern Ontario.

Appendix 8a

Major fish species and numbers taken at Ranger Lake during the summer of 1983 with wire minnow traps*

Lake No.	*Chrosomus* spp.	pearl dace	fathead minnow	brook stickleback	common shiner	creek chub	white sucker	blacknose dace	Iowa darter	yellow perch
	Fish species									
002R	–	1	–	–	1	–	–	–	–	45
003R	–	1	2	–	174	25	–	–	–	–
004R	–	233	–	–	331	188	18	14	–	–
005R	–	–	–	–	–	115	–	171	–	–
006R	–	–	2657	–	5	–	–	–	–	–
007R	–	–	3854	1	–	–	–	2	–	–
008R	–	–	510	–	–	148	–	48	–	–
009R	–	150	–	–	–	3	–	–	–	–
010R	1	1061	–	–	–	–	1	–	–	–
012R	–	–	–	10	–	–	–	3	–	–
021R	133	262	71	1	–	–	–	–	–	–
022R	135	–	–	–	15	–	4	3	–	–
025R	2	–	8	–	–	–	–	–	–	–
028R	–	–	–	–	66	–	3	–	–	–
029R	129	3	244	1	–	–	1	–	–	–
030R	–	36	–	–	84	4	3	–	–	–
033R	1270	13	–	1	–	–	2	–	–	–
036R	52	154	–	–	–	21	5	3	–	–
044R	–	8	–	–	–	–	–	–	–	–
048R	–	–	2292	–	–	213	–	3	–	–
051R	247	–	–	32	–	–	–	–	–	–
052R	–	63	–	–	–	–	–	–	–	–
058R	241	183	13	9	51	–	2	–	1	–
059R	101	–	–	12	3	1	–	7	–	–
060R	–	–	–	4	–	31	–	16	–	–
064R	2	–	–	–	–	2	–	–	–	–
065R	2541	–	–	–	–	–	–	–	–	–
079R	13	–	–	–	–	–	–	–	–	–
086R	377	–	–	–	4	9	2	30	–	–
087R	67	2	221	–	1	16	–	1	–	–
401R	1165	–	–	–	–	–	–	–	–	–
403R	498	21	330	25	–	–	3	–	–	–
404R	1523	–	–	–	–	–	–	–	1	–
405R	547	–	–	–	–	–	–	–	1	–
406R	220	–	–	26	4	57	2	–	–	–
407R	73	1	6	17	–	–	6	–	–	–
408R	147	1	81	16	39	11	9	–	–	–
409R	36	–	–	16	1	–	16	–	1	–
410R	472	691	11	17	–	1	50	–	–	–
414R	536	34	15	11	–	–	–	–	–	–
415R	566	4	4	10	–	2	9	–	–	–
417R	94	–	5	9	–	–	8	–	–	–
422R	43	2	–	–	–	1	–	–	–	–
423R	509	182	–	9	14	–	2	–	–	–
424R	1	1	–	1	336	6	10	1	–	–
426R	7	5	–	10	–	–	–	–	–	–
430R	326	–	–	–	–	–	–	–	–	–
431R	6	1	6	–	1	1	2	–	–	–
432R	461	14	–	–	–	1	1	–	–	–
441R	2340	9	–	3	–	–	–	–	1	–
445R	1672	103	–	–	–	–	–	–	5	–
447R	39	–	5	12	130	45	3	–	–	–
452R	29	–	–	–	387	29	21	–	–	–
453R	60	–	–	–	–	–	–	–	–	–
454R	290	–	–	–	–	–	–	–	3	–
456R	369	36	1	–	136	19	24	2	–	–
458R	2	2	–	8	18	1	22	7	–	–
464R	38	6	–	6	6	1	1	–	–	–
466R	179	689	–	–	–	–	–	–	–	–
484R	1770	–	–	–	–	–	2	–	–	–
488R	1666	–	–	–	–	–	–	–	–	–

* Additional species taken were brook trout (008R, 086R, 044R), golden shiner (059R, 086R), lake chub (034R) and blacknose shiner (058R). Lakes without fish were 013R, 014R, 015R, 019R, 020R, 420R, 436R and 459R.

Appendix 8b

Major fish species and numbers taken at Wanapitei during the summer of 1983 with wire minnow traps*

Lake No.	*Chrosomus* spp.	pearl dace	fathead minnow	brook stickleback	common shiner	creek chub	white sucker	blacknose dace	Iowa darter	yellow perch	rock bass	pumpkinseed sunfish	golden shiner
	Fish species												
002W	420	4	479	2	–	–	–	–	1	–	–	–	–
003W	–	–	–	–	–	–	–	–	1	–	–	–	–
005W	87	–	–	–	–	–	–	–	–	–	–	–	–
022W	–	–	–	78	–	–	–	–	–	–	–	–	–
240W	–	–	–	–	–	–	–	–	–	53	–	–	–
250W	–	–	–	–	–	–	–	–	–	10	–	–	–
252W	–	–	–	–	–	–	–	–	5	–	–	–	–
253W	–	–	–	–	–	–	1	–	–	43	–	–	–
254W	–	–	–	–	–	–	–	–	–	27	–	–	–
268W	–	–	–	–	–	–	–	–	–	15	–	–	–
333W	–	–	–	–	–	–	2	–	–	–	56	–	–
404W	–	–	–	–	–	–	–	–	–	5	–	–	–
409W	492	198	1505	–	–	–	6	–	1	–	–	–	–
475W	–	–	–	–	–	–	–	–	–	20	8	20	2
479W	1	–	–	–	–	–	–	–	–	–	–	–	–
480W	–	–	–	–	–	–	–	–	–	51	–	7	–
494W	–	–	–	–	–	–	–	–	–	24	–	3	–
501W	–	–	–	–	–	–	–	2	–	42	–	7	13
502W	–	–	–	–	–	–	–	–	–	–	6	1	–
515W	–	–	–	–	–	–	–	–	–	21	–	30	2
524W	–	–	–	–	–	–	71	–	–	–	–	–	262
526W	–	–	–	–		–	–	–	–	57	–	–	–
579W	–	197	–	–	–	–	–	–	–	–	–	–	–
583W	–	–	–	–	–	–	–	–	–	1	–	–	–
856W	1929	–	1	4	–	–	–	–	1	–	–	–	–
900W	1258	–	705	2	–	–	–	–	4	–	–	–	–
903W	421	368	70	5	284	–	68	–	–	–	–	–	–
904W	162	–	62	1	320	–	24	–	–	–	–	–	–
905W	–	55	–	–	58	3	2	–	–	100	–	–	–
907W	1700	11	–	–	–	–	–	–	–	–	–	–	–
909W	3377	–	–	–	–	–	–	–	5	–	–	–	–

* Additional species taken were smallmouth bass (475W), northern pike (515W), brown bullhead (256W) and an unidentified species (197W). Lakes without fish were 013W, 016W, 242W, 248W, 251W, 257W, 258W, 259W, 260W, 266W, 316W, 338W, 343W, 394W, 401W, 402W, 403W, 406W, 407W, 408W, 410W, 527W, 572W and 902W.

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